FANSHAWE LAKE:
THE NEED FOR WATER QUALITY MANAGEMENT IN SOUTHERN ONTARIO RESERVOIRS

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Report prepared for:

Liminology Section
Water Resources Branch
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ABSTRACT

Southern Ontario has the potential for the widespread construction and retrofitting of small reservoirs for hydroelectric power generation. As a consequence of the naturally fertile soils and intensive agricultural operations in much of southwestern Ontario, these reservoirs have become highly eutrophic and suffer from bottom water dissolved oxygen depletions and extensive blue-green algal blooms which degrade fish habitat and aesthetic values. Monitoring of hardwater reservoirs in the area indicates that bottom draw schemes for power generation are likely to release water of unacceptable quality. Data from Fanshawe Reservoir are presented as typical for these eutrophic hardwater systems, with Secchi depths of 3.5 to 0.4 m, an alkalinity of 207 to 92 mg/L as $CaCO_3$, chlorophyll *a* concentrations of 138 to 3 μg/L, euphotic zone total nitrogen at 9.22 to 1.07 mg/L as N and euphotic zone total phosphorus values of 0.138 to 0.044 mg/L as P. Methods to improve reservoir and tail water quality are discussed. Water quality management should become a major component of the management of reservoirs in southern Ontario.

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INTRODUCTION

Dams constructed on rivers and streams have created several surface water impoundments in southwestern Ontario. The primary purpose of these reservoirs is for spring flood control and summer flow augmentation; however, most also sustain intensive recreational use (swimming, boating, fishing). A series of reservoirs in southwestern Ontario have been monitored as part of the Ontario Ministry of the Environment.(M.O.E.) Inland Lakes program. The sites include Binbrook, Fanshawe, Gordon Pittock, Guelph, Springwater, Whittaker, and Woolwich Reservoirs. Fanshawe Reservoir is an example of a typical eutrophic hardwater flood control reservoir in the region. It was constructed in 1953 as a surface discharge multi-purpose reservoir. In 1963, a low level discharge (bottom water draw) began operating (Johnson & Berst 1965). Fanshawe Reservoir's seasonal trends of phytoplankton, zooplankton and water chemistry parameters were held in common with the other six reservoirs studied. Fanshawe Reservoir has rather severe dissolved oxygen problems related to eutrophication. Bottom water anoxia and internal loading of phosphorus via sediment feedback are widespread problems in these systems.

Fanshawe Reservoir is one of several reservoirs in the southwestern region of the province with a hydroelectric capacity. Others include Maple Hill (Rocky Saugeen River), Thornburry (Beaver River), Eugenia Falls (Bighead River), Scone Dam (North Saugeen River) and Williamsford Dam (North Saugeen River) (D. Loftus pers. comm.). Anticipated projects include Jancal Dam (Rocky Saugeen River), Cendeman Dam (Beaver River), Ayton Dam (South Saugeen River), McClure Dam (North Saugeen River) and Mooresburg Dam (North Saugeen) (D. Loftus, pers. comm.). A longer list of potential hardwater hydroelectric

plants may exist, but the authors were not able to get more extensive information at this time.

The Ontario Ministry of Energy has been encouraging the development of private hydro-electric facilities since 1981 (Veal, pers. comm.). Retrofitting of this type may become more widespread as pressure for alternate sources of electrical power mount in the future. The recently published Ontario Hydro Demand/Supply Report (1989) states that economic development of hydraulic resources are a high priority both on large scale (Hydro owned) and small scale (non-utility generation) projects. Small scale projects are described in the report as likely to be developed on rivers that already have dams or control structures not presently used for power generation. Southern Ontario also has many potential sites for new small scale hydro electric dams for power generation (M.N.R. 1985). For example, the Grand River has 15 sites with the potential to deliver over 50 KW 95% of the time. The Saugeen River has 16 such sites. Similar sites can be found on the Thames and Nottawasaga rivers, although many of these sites may not be viable for economic or technical reasons (M.N.R. 1985). Ontario Hydro (1989) estimates that small hydraulic non-utility generation projects in Ontario may provide over 300 MW of economically attainable power by the year 2000.

The development of hydroelectric power in Southern Ontario has profound implications for water quality in the region's watersheds. During normal operation, the reservoirs in the region release surface waters downstream. The surface waters of these lakes almost always have acceptable concentrations of dissolved oxygen and inorganic chemicals (although quality may be severely impaired during periods of blue-green algal

blooms). The worst quality water (low dissolved oxygen, high hydrogen sulphide and ammonia) is found in the bottom waters of the reservoirs. It is standard practice to discharge mid level or bottom water from a reservoir through turbines for power generation. The reservoir becomes a potential source of contamination downstream once it's bottom water is released for hydropower (Zimmerman 1988).

PART A: WATER QUALITY IN 1988 AND 1989

MATERIALS AND METHODS

1. Water Chemistry, Phytoplankton and Zooplankton

Samples were collected once every two weeks from the site during May to September of 1988 and 1989. Grab samples for water chemistry were collected from the North Thames River by lowering two weighted, narrow mouthed (2.1 cm diameter) 1 litre glass bottles into the river from a bridge on the Clarke Side road. Care was taken not to lower the bottles into the stream bed and disturb the sediment. Lake samples were taken by boat at two stations: F1 and F2 (Fig. 1). When the reservoir was at its normal summer operating level, the water columns at stations F1 and F2 were 10 m and 3 m deep respectively.

Standard Secchi disk readings were taken during each visit to the lake. A YSI Model 58 dissolved oxygen and temperature meter was used to obtain temperature and dissolved oxygen profiles at each station. Oxygen measurements were verified with the azide modification of the Winkler titration technique (APHA et al. 1976) on samples of surface and bottom water. Water samples were collected one meter off bottom (1 MOB) using a 6 litre PVC Van dorn bottle. Euphotic zone composite samples were taken by lowering and raising two weighted, narrow mouthed (2.1 cm diameter) 1 litre glass bottles through the euphotic zone so that they filled just as they reached the surface. The euphotic zone was defined as twice the observed Secchi disc depth to a maximum depth of 1 meter off lake

bottom. A portion of the euphotic zone sample was removed (500 mL) for phytoplankton enumeration (preserved with 2 mL of Lugol's solution). Another 500 mL of composite sample was removed for chlorophyll analyses and buffered with 2 mL of a 2% by weight $MgCO_3$ solution (to help prevent degradation of chlorophyll pigments). Two subsamples (500 mL each) were taken from the 1 MOB and composite water samples for chemical analyses and trace metal analyses. The trace metal subsamples were preserved with 1 mL of concentrated nitric acid to prevent precipitation.

Zooplankton samples were taken with a Clarke-Bumpus net (80 μm mesh, 12.3 cm diameter aperture) adapted for short vertical hauls with a sensitive flow meter. For each sample, the net was towed vertically from 1 MOB to the surface and then a second haul was made over the same distance with the net removed in order to calculate collection efficiency. Zooplankton samples were preserved with 4% sugared formalin.

All water chemistry analyses were performed by the Ontario Ministry of the Environment Rexdale Laboratory using their standard methods (MOE 1990a). Phytoplankton analyses were made using settling chambers and inverted microscopes for counting and obtaining dimensions. Samples for 1989 phytoplankton analyses were recombined or pooled to provide a single representative sample for the sampling season. Volume formulae used by the Aquatic Plant Unit were used to create biovolume data (MOE 1990b). Zooplankton analyses were performed using the ZEBRA automated system to calculate length vs. weight relationships. The method is described in Yan & Mackie (1987).

2. Calculation of Nutrient Loading

The STLOAD mass balance calculation program (Dorset Laboratory, MOE) was used to calculate the summer-time mass of nitrogen and phosphorus entering and leaving Fanshawe Reservoir. STLOAD links daily observations of water flow rates to irregular observations of water quality via linear interpolation. Inflow rates from the North Thames River were estimated from Environment Canada Station No. 02GD015 (Thorndale or Plover Mills). Outflow rates were estimated by comparing data from Environment Canada Station No. 02GD003 (just below stilling well of Fanshawe Dam) to known bottom and surface water discharges from the dam. Water quality data were taken from our collections at the North Thames inflow station and Station F1 in the reservoir.

RESULTS AND DISCUSSION

1. Upstream Water Quality

The North Thames River drainage basin includes most of Perth County. Land use in the basin is primarily agricultural. A variety of towns, however, are located directly on the river or its tributaries: Stratford (pop. 27,000), St. Marys (5,000), Mitchell (3,000) and Fullarton (200). As a result of this agricultural, industrial and urban development, the watershed has very high point source and non-point source nutrient inputs (Hayman 1989). The North Thames River experienced relatively strong flows well into June or even July (Fig. 2). The combination of high concentrations of nutrients and high flows created large

nutrient loadings to the reservoir. Dense growths of the green alga Cladophora (an indicator of eutrophic conditions) were observed throughout the ice free period along the river bed just upstream of the reservoir. The high loadings are reflected in the water quality data obtained from the grab samples collected at that site.

Fig. 3 shows a gradual increase in turbidity in the river water as the summers progressed with a drop in the fall. The pattern reflects flow rates in the river (Fig. 2). Low flow rates seem to correspond with more turbid water in the summer months. The anomalously high turbidity recorded on April 27, 1988, may be related to local construction activities observed in the area at that time. Iron values followed the turbidity pattern quite closely (Fig. 4). The link between iron and turbidity has been noted in inflow streams at other hardwater reservoirs in Ontario (Vandermeulen, unpubl. data) and it may be an indication of inorganic silt bound iron entering the system via erosion. The conductivity of the inflowing water was very variable (Fig. 5) and did not seem to follow flow patterns. The average conductivity seen (500 to 550 μS/cm) did fall within the range of the other rivers sampled in agricultural watersheds in southwestern Ontario (Vandermeulen, unpubl. data). Dissolved carbon values (Fig. 6) were also typical of more nutrient laden reservoir inflows. The bulk of the phosphorus carried by the North Thames river was in particulate form as indicated by the relatively high proportion of total phosphorus present compared to orthophosphorus (Fig. 7). Total phosphorus values were high and exceeded a general water quality guideline of 0.03 mg/L for rivers and streams (MOE, 1984). Kjeldahl nitrogen values were also high (Fig. 8), indicating the particulate nature of nutrients carried by the river. The ammonium values described in the same figure exceeded 0.05 mg/L on several occasions.

2. Reservoir Water Quality

As expected, Fanshawe reservoir responded to the nutrient loadings provided by the North Thames River. Fig. 9 shows how nitrate concentrations in the river were reflected within a short time lag at the surface waters near the dam 7 km away. The total nitrogen / total phosphorus ratios recorded for the river (Fig. 10) were followed by the same pattern in the reservoir (Fig. 11). The high nutrient loadings led to phytoplankton blooms, bottom water anoxia and sediment feedback of nutrients and metals as described below. All data are presented for station F1 only as very similar chemistry and plankton data were obtained at F2.

The volume of water held by the reservoir during the summer sampling periods in 1988 and 1989 was relatively constant at between 1.2 and 1.6 X 10^{10} litres. Major fluctuations in reservoir level were confined to the other months of the year (Fig. 12). Surface flood gates were routinely opened to allow epilimnetic water to escape the reservoir during periods of high inflow (Fig. 13, compare to Fig. 2). Bottom water outflow (water drawn from 9 m depth through a 60 inch low flow valve or turbine intakes) was maintained year-round (Fig. 14) and was the only outflow from the reservoir during the summer months (see Fig. 13).

Temperature and Dissolved Oxygen

The bottom water near the dam was only minimally colder than the surface waters during the summer (Fig. 15). For this reason, the term "bottom water" is used throughout this report rather than "hypolimnion". Poor stratification may have been due, in part, to wind mixing of surface waters as two other nearby flood control reservoirs with similar morphometry (Binbrook and Gordon Pittock mean depth approximately 3 m, surface area about 200 ha), were also poorly stratified (Vandermeulen, unpublished data). The weak thermocline may have also been partly due to the destratifying effect of bottom water draw for hydroelectric power generation (Johnson and Berst 1965). Even in the face of relatively poor stratification, bottom water anoxia developed rapidly during the summer (compare Figs. 16 and 17). Most of the bottom water had less than 4 ppm oxygen by the first week in June in both years of sampling (Figs. 18 and 19).

It should be stressed at this point that the decline in oxygen in the bottom waters represented a chemocline whose depth was somewhat independent of the weak thermocline observed. Even though thermal stratification was weak, it definitely did exist and the surface waters were not constantly and thoroughly mixed with the bottom waters. Indeed, temperature based density stratification (physical isolation) of waters can occur at temperature differences of 2°C or even less (Burns, unpubi. data). Fanshawe Reservoir did behave as a stratified dimictic lake with a fall overturn.

Bottom water anoxia was observed on a regular basis during the early years of operation of the dam. Bottom draw regimes in the past have, however, corrected the

problem to some extent (Johnson and Berst 1965). In 1988 and 1989, sediment oxygen demand (and bottom water oxygen demand) may have been greater as bottom draw no longer seemed to prevent anoxia in a large proportion of the bottom waters.

For example, a rare closure of the bottom draw valves occurred from June 1 to 19, 1989 (Fig. 14). As in 1988, the water near the bottom became anoxic by the end of the first week in June 1989. The closure of bottom draw in the first couple of weeks in June 1989 did not cause bottom water anoxia. It merely coincided with its normal development (Fig. 18). In any case, the reopening of bottom draw valves on June 20, 1989, as the exclusive release of water from the dam for the rest of the season (no epilimnetic water released through flood gates from June 20 to November 16, 1989), did not remove this anoxic water to any great extent. It remained until the fall overturn as in 1988 (Fig. 19).

A potentially catastrophic event was observed during both years of monitoring. Mixing during the start of fall overturn (early September) caused surface waters to drop below 5 ppm oxygen at station F1 (Fig. 20). The effect was seen as far up the reservoir as station F2 in 1989. If this fall mixing drops the water column dissolved oxygen much below 4 ppm, fish will not be able to survive in the lake.

Water Chemistry

Bottom water anoxia causes sediments to enter a reduced state which allows a variety of sediment bound hydroxide precipitates to dissolve. As a result, concentrations of several metal ions then increase in the bottom waters. Figs. 21 and 22 illustrate how manganese and iron concentrations increased in the bottom waters soon after anoxia happened in June. Water quality was worse in 1989 and the sediments released more manganese and iron in that year, as well greater amounts of other ions (see below). Orthophosphorus was released by the sediments as well. This is known as the phenomenon of internal phosphorus loading (Fig. 23).

It was particularly disconcerting to note that bottom water containing substantial amounts of ammonium ion released by the sediments (or from anoxic metabolism within the bottom water column) became mixed with surface waters during fall overturn. The entire water column had over 1 mg/L ammonium on September 19, 1989 (Fig. 24). At the temperature and pH of the water at that time, approximately 4% or 0.04 mg/L of the ammonium would be found in the unionized form as ammonia. The MOE "Blue Book" states that "Concentrations of un-ionized ammonia should not exceed 0.02 mg/L for the protection of aquatic life" (MOE, 1984). In other words, the entire volume of the lake (including station F2) had over twice the concentration of ammonia considered "safe".

Low oxygen, coupled with high ammonia concentrations, have the potential to cause massive fish kills throughout the lake. If water quality in the reservoir becomes much worse, the fish mortalities may become a regular occurrence. A fish kill involving white suckers did

occur in July 1989. An analysis by the University of Guelph indicated that the fish had contracted a variety of diseases that were promoted by stressful water quality conditions, high water temperatures, high pH, high suspended solids and low dissolved oxygen (reports by J. Westwood and B. Hawkins, London MOE). Other fish kills were noted in 1988, see below.

A variety of other water quality parameters were monitored in the reservoir as well. Bottom waters were more turbid than surface waters (Fig. 25) as is typical for these eutrophic hardwater systems where bottom water is tea coloured due to accumulated organic matter. Conductivity values in the lake (Fig. 26) fell within the same range as the inflow waters (Fig. 5). Dissolved inorganic carbon concentrations were lower in the euphotic zone due to photosynthetic uptake at the surface and anoxic release at the bottom (Fig. 27). Organic carbon values peaked in surface waters due to dense phytoplankton blooms, particularly in 1989 (Fig. 28). Total phosphorus and Kjeldahl nitrogen values were high in the euphotic zone because of particulate phosphorus and nitrogen associated with phytoplankton. Total phosphorus and Kjeldahl nitrogen values were even higher in the bottom waters because of the internal loading mentioned earlier (Figs. 29 and 30).

Phytoplankton

Phytoplankton growth followed the trends in water quality parameters mentioned above. Water quality was worse in 1989 and phytoplankton densities were also far higher. The clarity of the water, as measured by Secchi disc, (Fig. 31) was between 1 and 2 m in 1988 and dropped to between 1 and 0.5 m during the dense algal blooms of 1989. The

sudden increase in surface water clarity (Secchi depth) in September 1989 was due to a crash or collapse of the phytoplankton bloom on the lake (see also Fig. 25). The crash in phytoplankton numbers would also add to the very low oxygen levels seen in the lake during that month. The peak chlorophyll a values of over 130 μg/L seen in 1989 (Fig. 32) were amongst the highest recorded in any of the monitored hardwater lakes (unpublished data). The pronounced crash in the phytoplankton community by September 1989 was reflected in the chlorophyll a values as well (Fig. 32)...

A more detailed analysis of the phytoplankton data for 1988 indicates a spring bloom in biomass followed by a late summer bloom (Fig. 33). The spring bloom was comprised almost entirely of diatoms (Figs. 34 and 35). The main genera were Stephanodiscus and Cyclotella. Green algae (mainly Chlamydomonas, Gloeocystis and Oocystis) and members of the Cryptophyceae (primarily Cryptomonas) predominated in the mid-summer while dinoflagellates (Ceratium) and the Cyanophyta (Aphanizomenon and Anabaena) accounted for a large proportion of the late summer bloom (Figs. 34 and 35).

The 1989 recombined phytoplankton sample analysis is shown in Fig. 36. Note that the average total algal biovolume in 1989 was over 10 mm^3/L, a value only seen during peak spring and late summer blooms in 1988 (Fig. 33). The seasonal species composition was basically the same in both years, with the exception that Oscillatoria became abundant along with Aphanizomenon and Anabaena in the late summer bloom of 1989, while Ceratium was not seen.

Water quality conditions have been relatively poor in Fanshawe Reservoir for at least the past two decades. Work done by David Osmond in 1973 (unpublished) describes bottom water anoxia (even though bottom draw was in effect in the summer of 1973) and his values for iron, orthophosphorus, ammonium, total phosphorus and total nitrogen in surface and bottom waters are similar to those presented here. He does mention past blooms of blue green algae on the lake, although in 1973 he found chlorophyll a values of only 4.6 to 15.5 μg/L (compare to Fig. 32), the dominance of the green alga Scenedesmus and abundant zooplankton. A report by Robinson (1973) corroborates some of Osmond's results.

Fanshawe Reservoir is unusual (but sadly not unique) in having a history of beach closures caused by the dense blue-green algal blooms (Fig. 37) (Hayman 1989). It has been estimated that the gross value of lost income to the Conservation Authority due to severe algal blooms on the lake is about $48,000 per year. Approximately $10,000 per year in gross gate fees are directly lost due to undesirable algae related poor lake aesthetics (Molot & Usher 1987). Beaches are often closed by the Ontario Ministry of Health due to elevated bacterial counts. Indeed, bacteria at Fanshawe were enough of a problem that an enclosure with an ultraviolet sterilization system was tested in 1988 to provide a "clean" swimming area (Kelleher 1989). It is apparent that algae cause significant economic problems (due to loss of gate fees upon beach closures) and control methods need to be developed and implemented.

Zooplankton

Zooplankton abundance varied in an interesting manner in relation to algal composition and density. In 1988, zooplankters were most abundant when total algal biovolumes were lowest in early summer (compare Fig. 38 to Fig. 33). The inverse relationship suggests herbivore control of phytoplankton density. The decline in zooplankton abundance in late June may have been due to predation by planktivorous fish, particularly young of the year (YOY) fish as evidenced by the declining proportion of large bodied Daphnia (Fig. 38). The establishment of anoxic bottom water at the same time would have hindered diurnal vertical migrations of the zooplankton to portions of the water column with less predation pressure and reduced the volume of habitat available. Soon after the zooplankton decline, phytoplankton began to bloom on the lake (Ceratium and the blue green genera).

The zooplankton were inhibited for much of the 1989 season (Fig. 38). This may have been due to poorer water quality in general in 1989. The lack of an active community of planktonic herbivores is thought to have contributed to the burgeoning algal populations seen in 1989. Zooplankton population densities did not increase until the late August decline in phytoplankton biomass (Fig. 33). It is significant that Daphnia accounted for most of that late season zooplankton biomass (Figs. 38 and 39). Active herbivory by Daphnia may have been partially responsible for the decline in algal biomass. A similar late season pattern was seen in 1988 (Fig. 38).

Tail Water Quality

Water leaving the Fanshawe Reservoir was checked on occasion by regional MOE staff. Low oxygen problems were measured by MOE staff and the City of London near the dam. A report by J. Westwood (MOE, Water Resources Assessment Unit, London) describes how black anoxic water smelling of hydrogen sulphide was released by the hydro electric power generator discharge at the base of Fanshawe Dam on August 11, 1988. It was the only discharge leaving the dam at the time. A fish kill had just occurred near the discharge involving large common white suckers and gizzard shad. Dead white suckers were also found along the shore of the reservoir near the dam. Bass, perch and walleye were also killed (D. Veal). Hundreds of crayfish were also seen crawling out of the lake onto shore.

3. Mass Balance Calculations

The results of the STLOAD calculations are presented in Table 1. The calculation of reservoir volume is approximate. For this reason, the differences of outflow and inflow (column C) do not exactly correspond to storage changes (column D) in Table 1. During the summer of 1988, the reservoir consistently stored total phosphorus (all values negative in column A). The storage effect could be explained as a net retention of water by the reservoir during that summer (all values negative in column C) because it was a very dry year. In 1989 however, the reservoir behaved in exactly the opposite manner for phosphorus. It consistently acted as a source of phosphorus (all values positive in column A). The 1989 behaviour could be explained partly to some net flushing of the reservoir in July and August (positive values in column C for those two months) but massive internal loading of

phosphorus during that summer coupled with bottom water discharge are far more likely reasons for the net export of phosphorus from the system. For example, phosphorus left the reservoir in excess of total input even in the face of a net retention of water in May, June and September of 1989 (Table 1). The bottom water being discharged in 1989 had double the concentration of total phosphorus compared to 1988 releases (Table 2). The release of phosphorus from Fanshawe Reservoir during the summer months is concentration driven (internal loading) rather than flow driven (flushing).

The mass balance for total nitrogen passing through the reservoir in 1988 and 1989 is more complex than that for total phosphorus. In general, 1989 had a summer of net release of nitrogen, likely due to internal loading (Table 1 column B). A series of loading calculations on the components of total nitrogen (ammonium, nitrate, organic N) may provide a clearer description of the movement of nitrogen through the system in the future.

PART B: OPTIONS FOR WATER QUALITY ENHANCEMENT

1. Upstream

The most direct and effective method to improve water quality in Fanshawe Reservoir is by controlling upstream point source (PS) and non-point source (NPS) pollution (Fig. 40). An excellent discussion on this form of cleanup in the Fanshawe Reservoir watershed is provided by Hayman (1989). For reducing phosphorus loadings, point source and erosion control should be major components of this effort (Hayman 1989). Turbid water and siltation was seen in much of the upper basin of Fanshawe Reservoir. Newly deposited sediment has been observed as far downstream as station F2.

A reduction in nutrient loading from upstream sources will be reflected in less dense algal blooms on the lake and a long term development of higher oxygen content in the bottom waters. It is encouraging that point source controls have recently been employed in the watershed (Hayman 1989).

A substantial amount of erosion occurs along the steep sides of the reservoir itself. Sodding and tree planting could help to control this on site source of nutrients.

2. In Reservoir

A variety of methods have been developed in the United States and Europe for improving water quality at a whole lake level. The journal Lake and Reservoir Management specializes in documenting these research efforts. One accepted method for improving bottom water oxygen levels is destratification (the mechanically induced mixing of a water column). A propeller driven destratification unit which draws mid-level water to the surface has been developed by the Aquatic Plant Unit at the Rexdale Laboratory (M.O.E.) and has been successfully operated at Guelph Lake (Vandermeulen, unpublished). It is estimated that at least five units would be required to mix Fanshawe Reservoir, each one capable of moving water at a rate of 3 $m^3 \cdot s^{-1}$ (Fig. 41). Each unit would cost approximately $30K to build for a total cost of $150K. Electrical costs would exceed $5K / yr but using power from the turbines in the dam may result in some cost savings. Destratification will likely solve bottom water low oxygen problems, but may not control algae effectively.

The establishment of a sustainable population of walleye (Stizostedion vitreum vitreum) in the reservoir through transplantations (Hunter & Schraeder 1989, Schraeder & Hunter 1990) may eventually have some effects on phytoplankton density via food chain effects (e.g. Van Donk et al. 1989).

3. Tail Waters

Discharge water (tail water) quality from Fanshawe Reservoir could be improved by pushing surface water with acceptable dissolved oxygen concentrations down into the hypolimnetic zone for release (Fig. 42). Holland (1984) provided design parameters for such a system. Another method to improve tail water quality is selectively withdrawing water at specific levels in the reservoir known to have good dissolved oxygen content (Fig. 43). Turbine venting (Fig. 44) can be retrofitted on existing hydroelectric equipment to add oxygen just prior to release (Bohac et al. 1986). A venturi on an existing discharge pipe may serve to elevate oxygen levels in a similar manner (Fig. 45). A system of baffles and weirs downstream of the discharge may agitate the tail waters sufficiently to aerate as well (Fig. 46).

PART C: LEGAL IMPLICATIONS

1. The U.S. Experience

In January 1982, the National Wildlife Federation filed a lawsuit against the U.S. Environmental Protection Agency stating that reservoirs were in effect point source discharges of pollutants (Attey & Liebert 1986). The District Court for the District of Columbia ruled in favour of the Federation in April of the same year. The EPA and "a multitude of electrical companies" (Attey & Liebert 1986 page 196) appealed and the ruling was overturned by the Circuit Court of the District of Columbia in November 1982. Attey & Liebert (1986) hold that the Circuit Court decision was erroneous and a serious threat to the "economic viability and biological integrity of the nation's waterways" (p. 195).

Three American Federal agencies are responsible for reservoir construction and operation: the U.S. Bureau of Reclamation, the Tennessee Valley Authority and the U.S. Army Corps of Engineers. All three agencies presently, either voluntarily or through legislation, regard water quality issues as a top priority. Water quality is considered to be as important as other purposes and outcomes of reservoir use. A special session of the 8th Annual International Symposium on Lake and Watershed Management was devoted to the topic (Timblin et al. 1988, Ruane et al. 1988, Buelow & Lamar 1988). For example, the Water Resources Development Act of 1986 states that benefits associated with environmental quality features that are justified and included in a proposed project shall be considered to at least equal the cost of those features so as to not depress or distort the

benefit-cost ratio. This provides recognition that water quality and other environmental factors have economic value (Buelow pers. comm.).

2. Potential Regulations in Ontario

The water quality problems associated with Fanshawe Reservoir have legal implications. Downstream fishkills attributed to discharge water with low dissolved oxygen concentrations have the potential to generate legal action against the Conservation Authority (D. Veal Pers. Comm.).

Hopefully, water quality problems associated with reservoirs can be resolved in a negotiated manner without resorting to legal action. An example is the history of the Scotch Block Reservoir. The reservoir was designed as a bottom draw system and was completed on 16 Mile Creek (Oakville Creek) in 1971 (MacBeth 1973). Unacceptable tailwater quality (anoxia) was noted in 1972. Acting upon a request from the M.O.E., the Halton Region Conservation Authority voluntarily installed an air driven destratification system in the reservoir in July 1973 (MacBeth 1973). Tail water quality improved soon afterwards.

At present, no Federal or Provincial legislation exists to specifically link water quality with reservoirs in Ontario (J. Herlihy pers. comm.). If legal action, however, does become necessary as an outcome of reservoir operation, a variety of existing legislation can be applied. The Ontario Environmental Assessment Act can be used to ensure that new projects (or retrofits of existing dams) have been evaluated for potential environmental impacts before construction begins. The Ontario Water Resources Act and Environmental

Protection Act can be used to prosecute in cases where discharges from existing dams are found to pollute. The Ontario Ministry of Natural Resources has the authority to apply the Federal Fisheries Act and the Ontario Lakes and Rivers Improvement Act on dam related issues.

A recent deliberate discharge of accumulated silt from a reservoir on the Rocky Saugeen River in preparation for conversion to hydroelectric power generation may serve as an example. The M.O.E. held that the silt was a contaminant deleterious to the environment and laid charges under Part 9 of the Ontario Environmental Protection Act for failure to report a spill of a contaminant. The M.N.R. laid charges under the Federal Fisheries Act and the Lakes and Rivers Improvement Act.

It may be possible to apply similar charges in cases where reservoir tail waters are found to contain high levels of ammonia and hydrogen sulphide with minimal oxygen content (J. Herlihy pers. comm.). Reservoir projects which incorporate hypolimnetic drawdown schemes, including hydroelectric projects, should be particularly sensitive to this type of legal action.

ACKNOWLEDGEMENTS

K.H. Nicholls (Water Resources Branch, M.O.E.) managed to keep research and sampling programs going at a variety of hardwater lakes and reservoirs long before the importance of such activity was formally recognized via the introduction of the M.O.E. Inland Lakes program. His work set the stage for the detailed sampling described herein.

D. Veal (Southwestern Region, M.O.E.) has consistently been supportive of our activities on Fanshawe Reservoir. He provided advice and information on the section dealing with legal implications. J. Herlihy (Legal Services Branch, M.O.E.) kindly described the Rocky Saugeen case and its application to reservoir discharges. D. Buelow (U.S. Army Corps of Engineers) helped to interpret the legal aspects of reservoir releases in the United States and provided exact dates for the NWF vs EPA case.

D. Loftus (Owen Sound District, M.N.R.) provided a list of microhydro projects in his district. D. Hayman (Upper Thames River Conservation Authority) gave inflow, outflow and storage capacity data for Fanshawe Reservoir from 1987 to 1989. N. Hutchinson (Dorset Laboratory, MOE) ran the STLOAD program on the flow and water chemistry data.

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Table 1. Summary of Mass Loading Calculations for Fanshawe Reservoir

Date	A Outflow minus inflow Total P (kg X 10^3)	B Outflow minus inflow Total N (kg X 10^3)	C Outflow minus inflow volume (l X 10^9)	D Average Reservoir storage volume (l X 10^9)
May 1988	-0.26	-75	-7.6	12.8
June	-0.09	+6.6	-1.5	12.2
July	-0.17	+0.63	-1.7	12.4
Aug.	-0.03	-0.16	-0.7	12.3
Sept.	-0.10	-2.1	-3.3	14.0
May 1989	+0.63	+10	-3.6	12.9
June	+0.92	-40	-1.5	13.8
July	+0.40	+16	+0.16	13.4
Aug.	+0.60	+7.1	+1.3	13.7
Sept.	+0.17	+0.90	-1.6	14.0

Table 2. Average Total Phosphorus concentrations (mg/L) for the period May to September at the inflow and in the bottom waters of Fanshawe Reservoir

	1988	1989
North Thames Inflow	.060 (n=10)	.049 (n=11)
Station F1 Bottom Waters	.067 (n=10)	.138 (n=11)

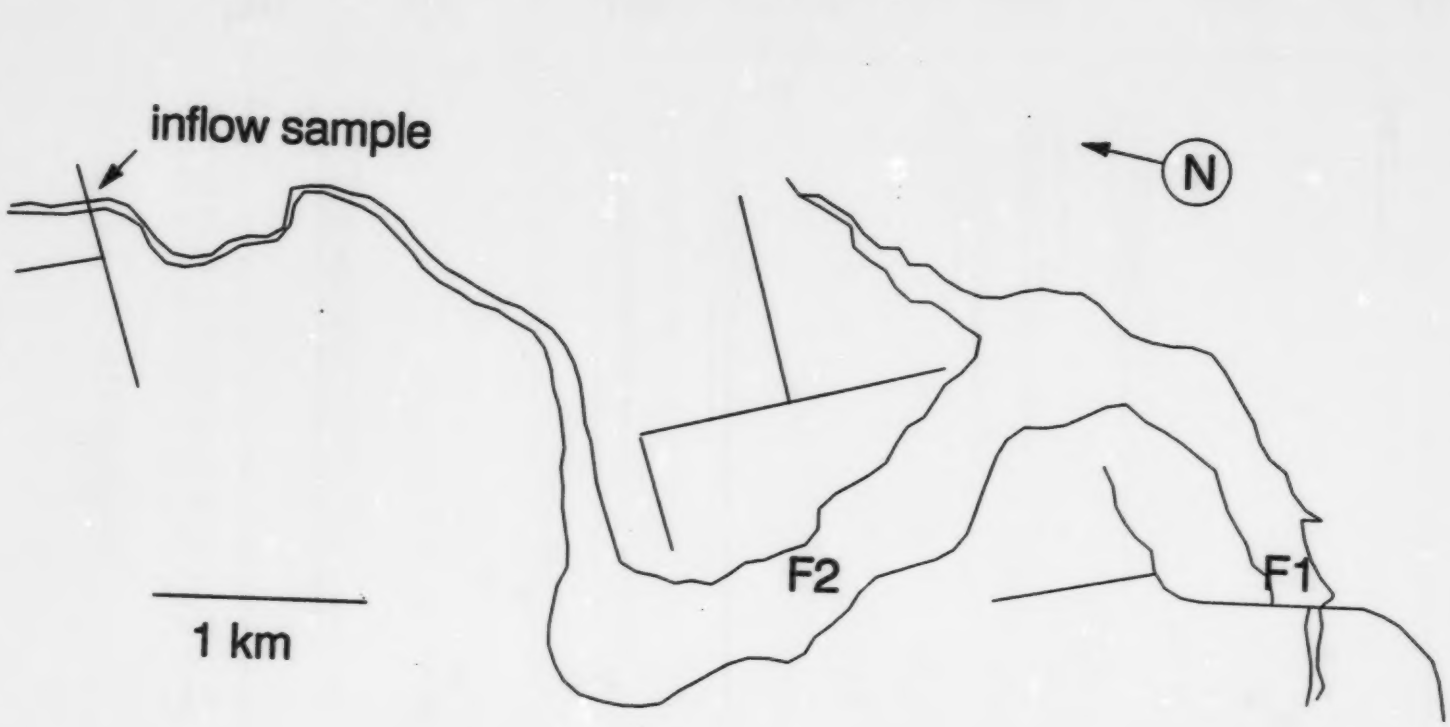


Fig. 1

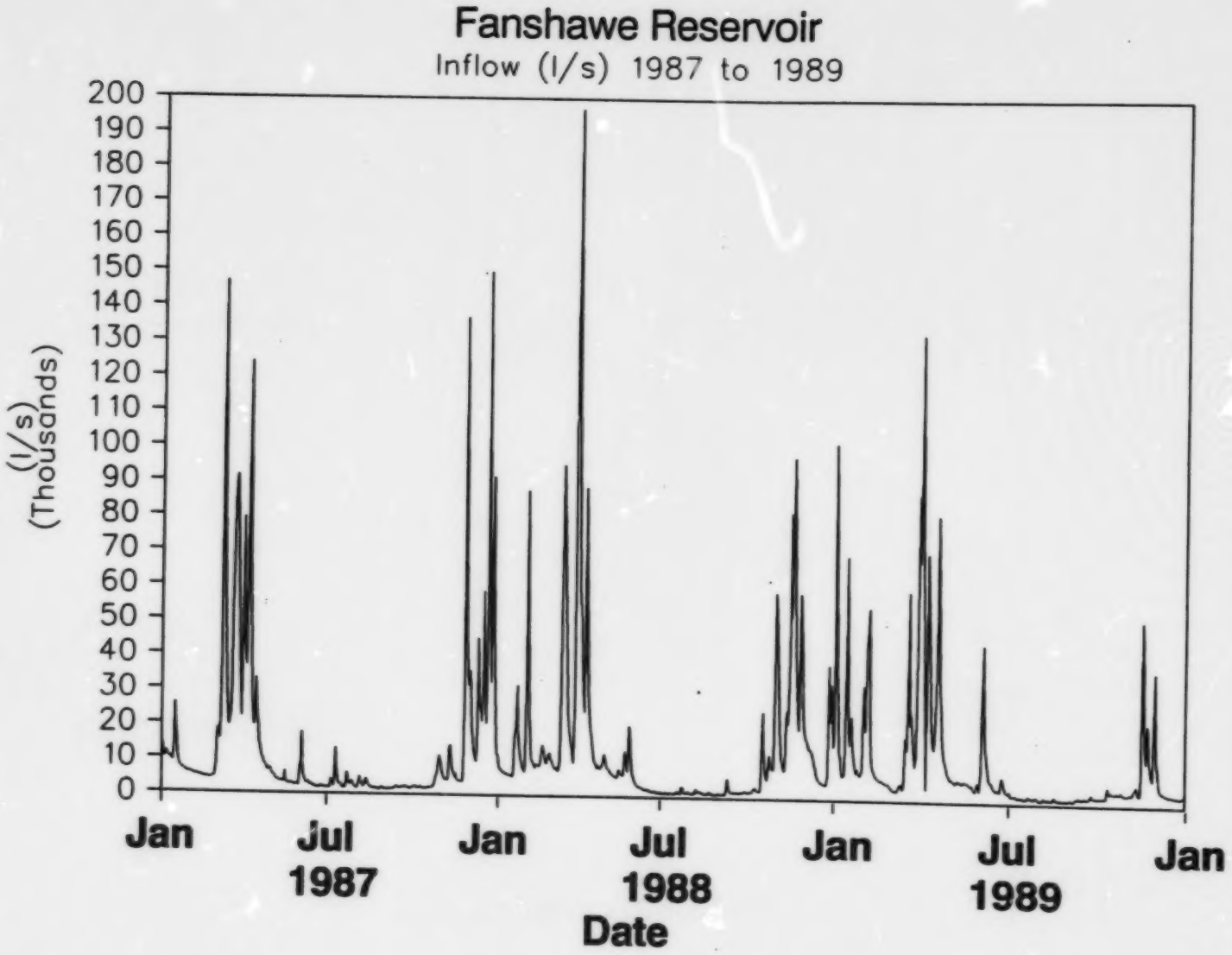

Fanshawe Reservoir
Inflow (l/s) 1987 to 1989
(l/s)
(Thousands)
200
190
180
170
160
150
140
130
120
110
100
90
80
70
60
50
40
30
20
10
0
Jan
Jul
1987
Jan
Jul
1988
Jan
Jul
1989
Jan
Date

Fig. 2

Thames River Inflow to Fanshawe Reservoir

Turbidity (FTU)

1988/89

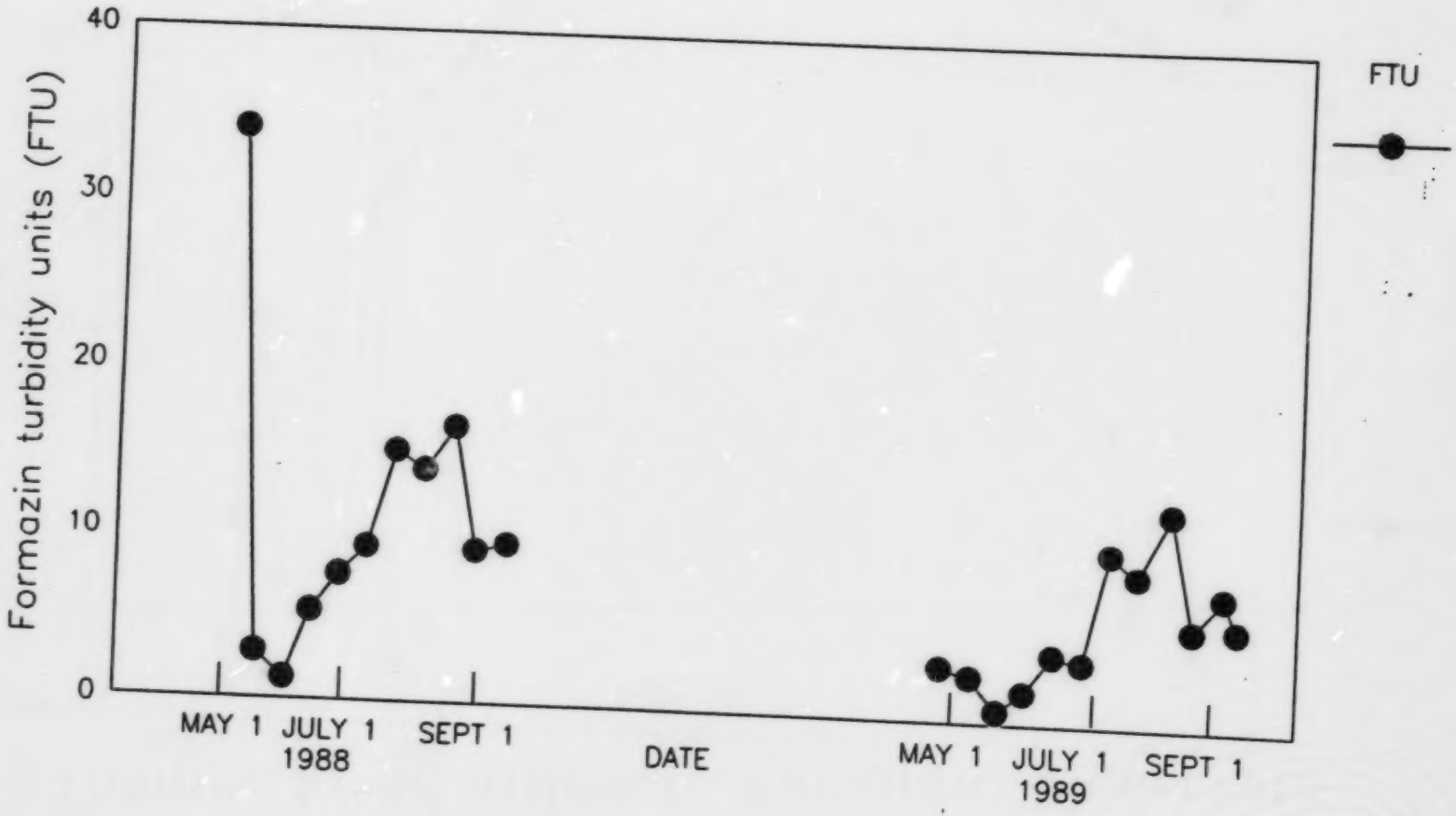


Fig. 3

Thames River Inflow to Fanshawe Reservoir

Iron

1988/89

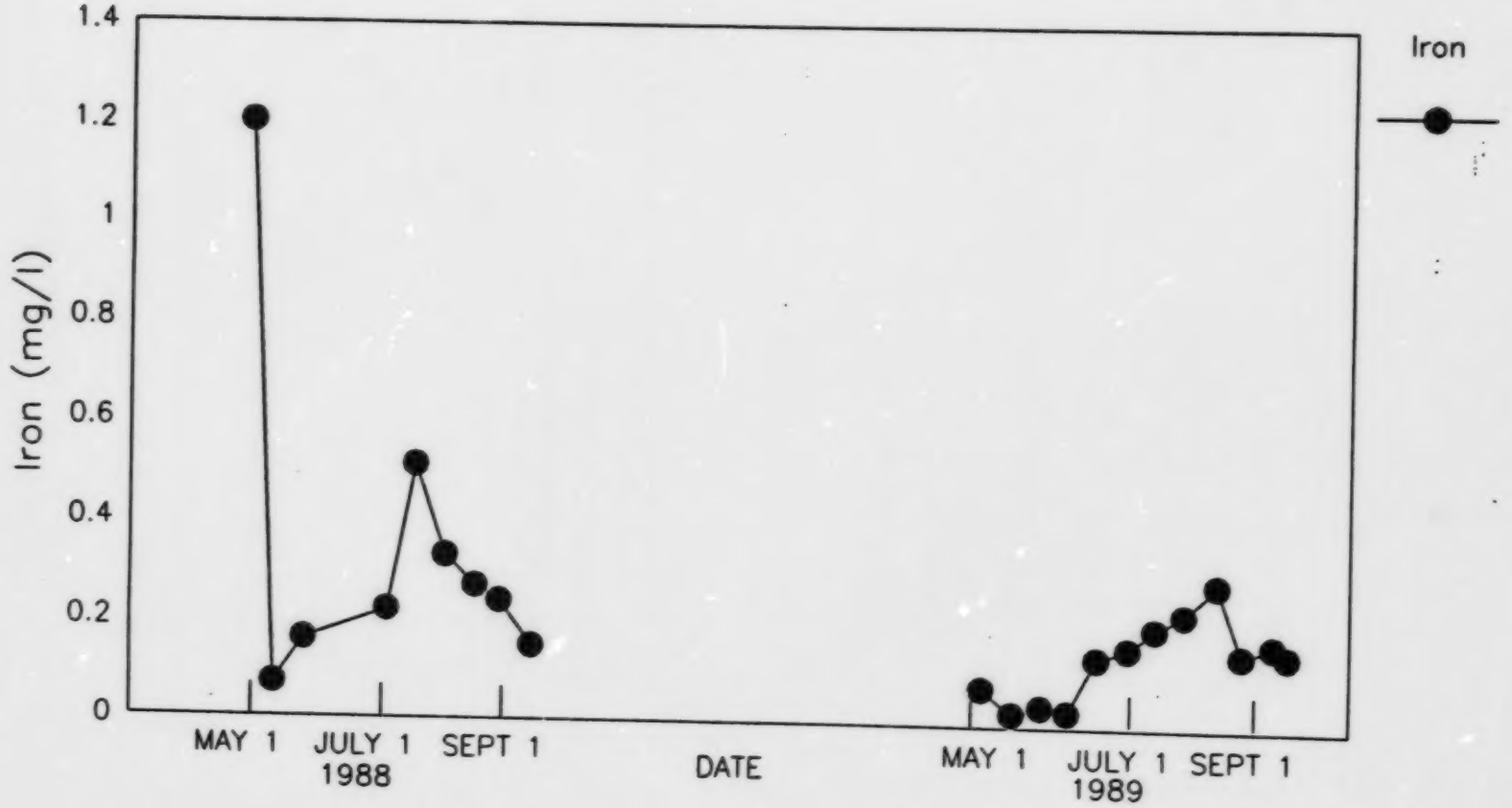


Fig. 4

Thames River Inflow to Fanshawe Reservoir

Conductivity (µS/cm)

1988/89

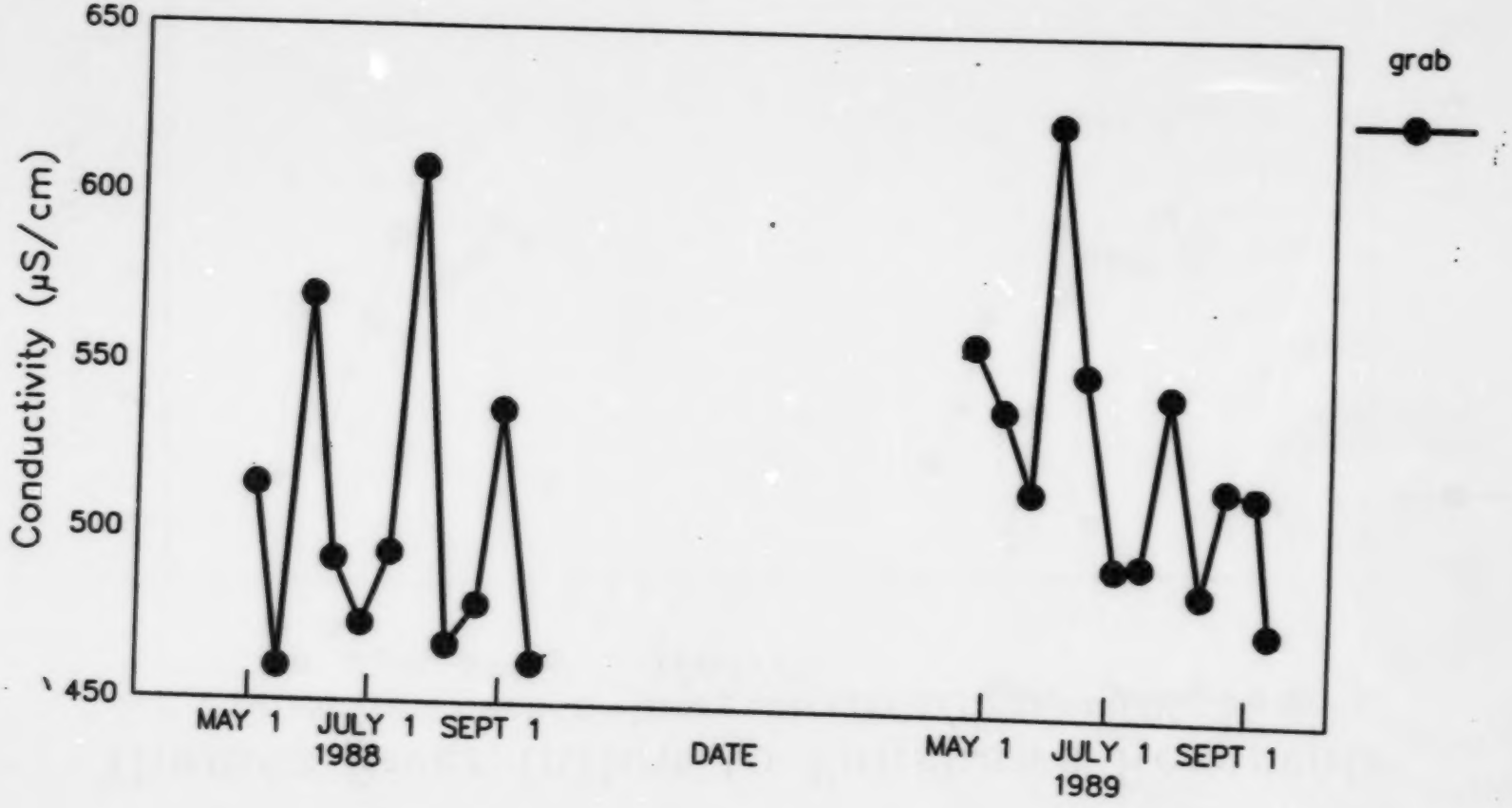


Fig. 5

Thames River Inflow to Fanshawe Reservoir

Dissolved Carbon–Inorganic(DIC)/Organic(DOC)
1988/89

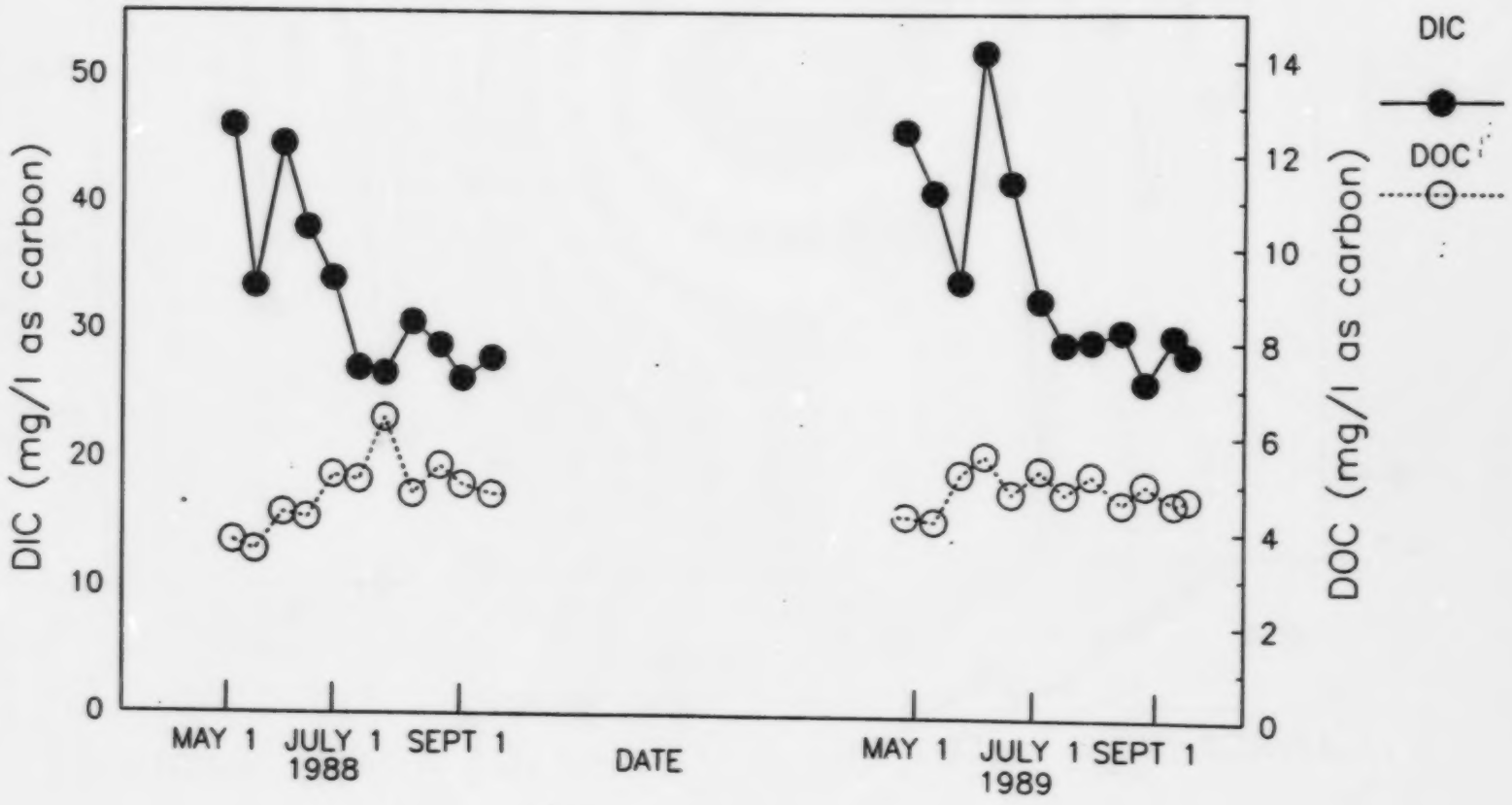


Fig. 6

Thames River Inflow to Fanshawe Reservoir

Total Phosphorus / Soluble Reactive Phosphorus
1988/89

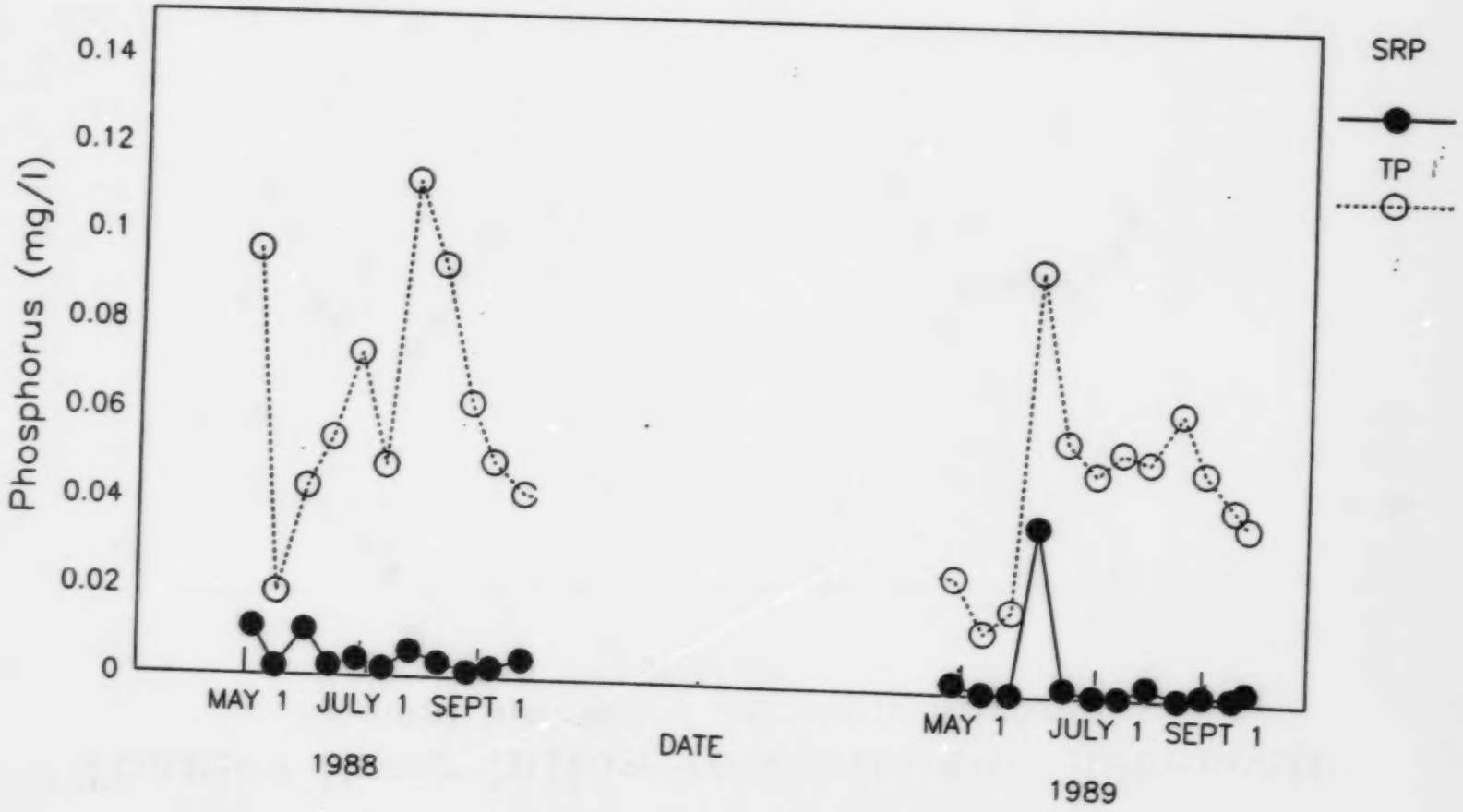


Fig. 7

Thames River Inflow to Fanshawe Reservoir

Kjeldahl Nitrogen / Ammonia Nitrogen

1988/89

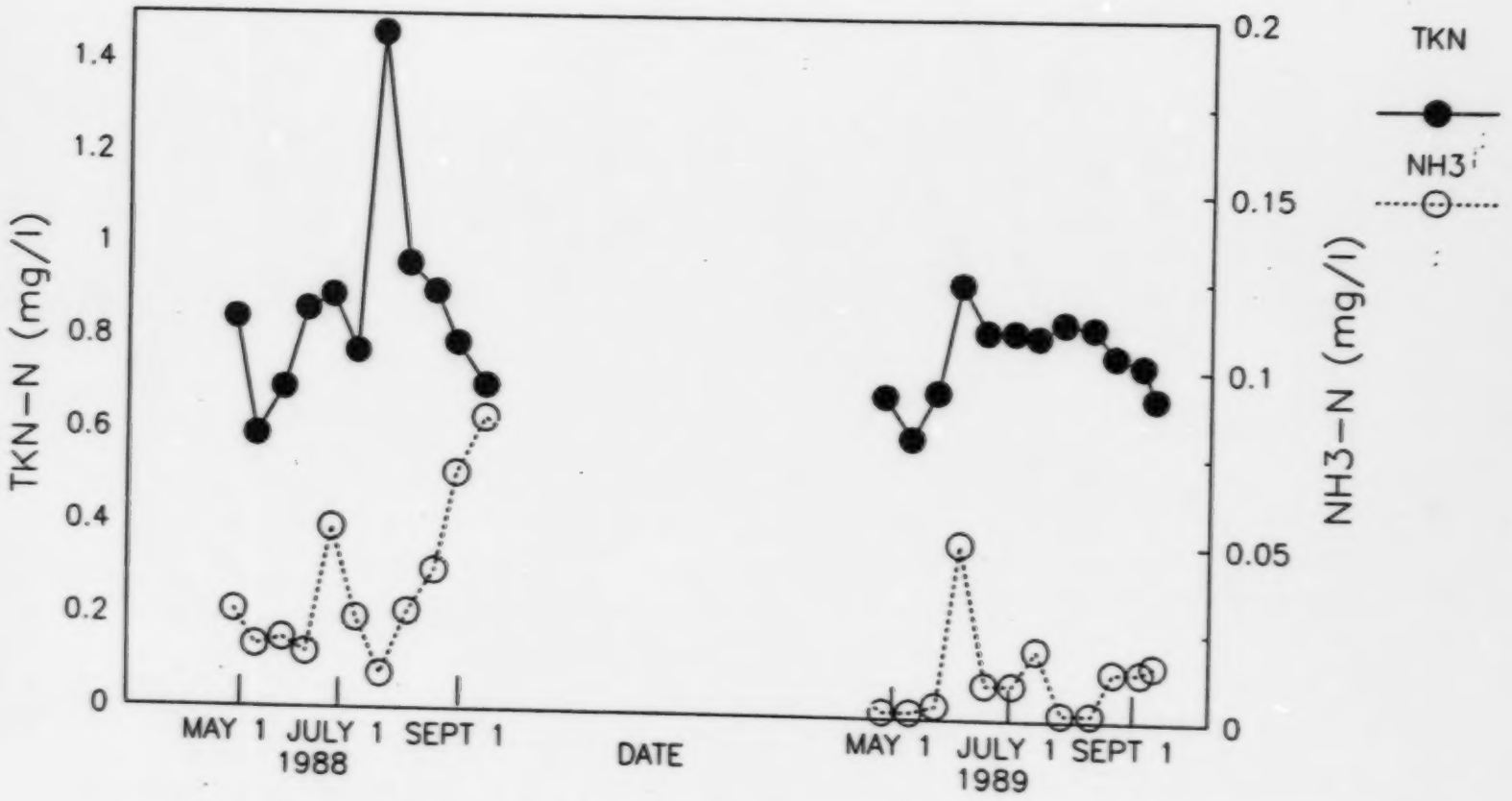


Fig. 8

Fanshawe Reservoir / Thames River Inflow

Total Nitrates (NO3 + NO2)

1988/89

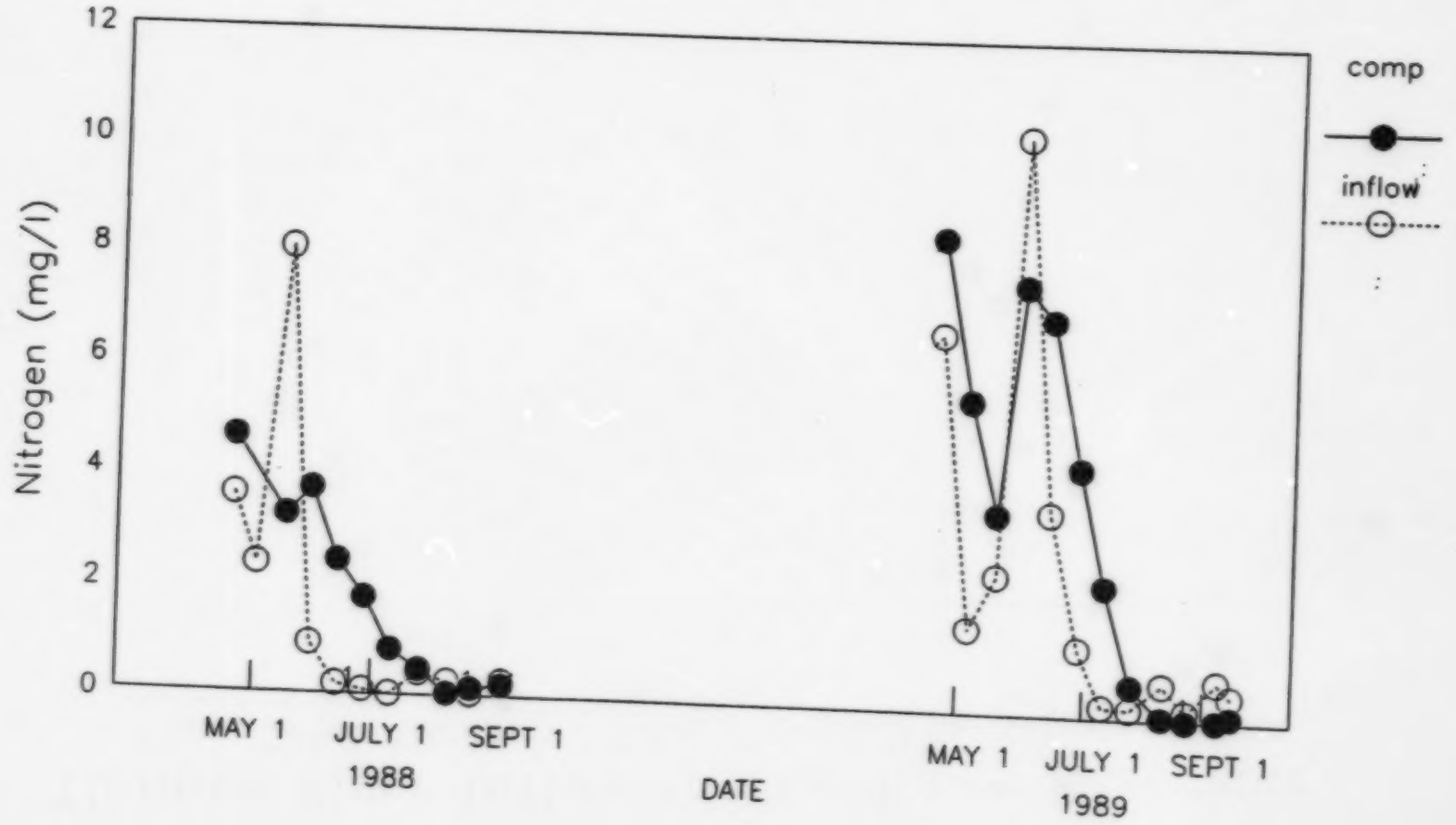


Fig. 9

Thames River Inflow to Fanshawe Reservoir

Total Nitrogen : Total Phosphorus Ratio

1988/89

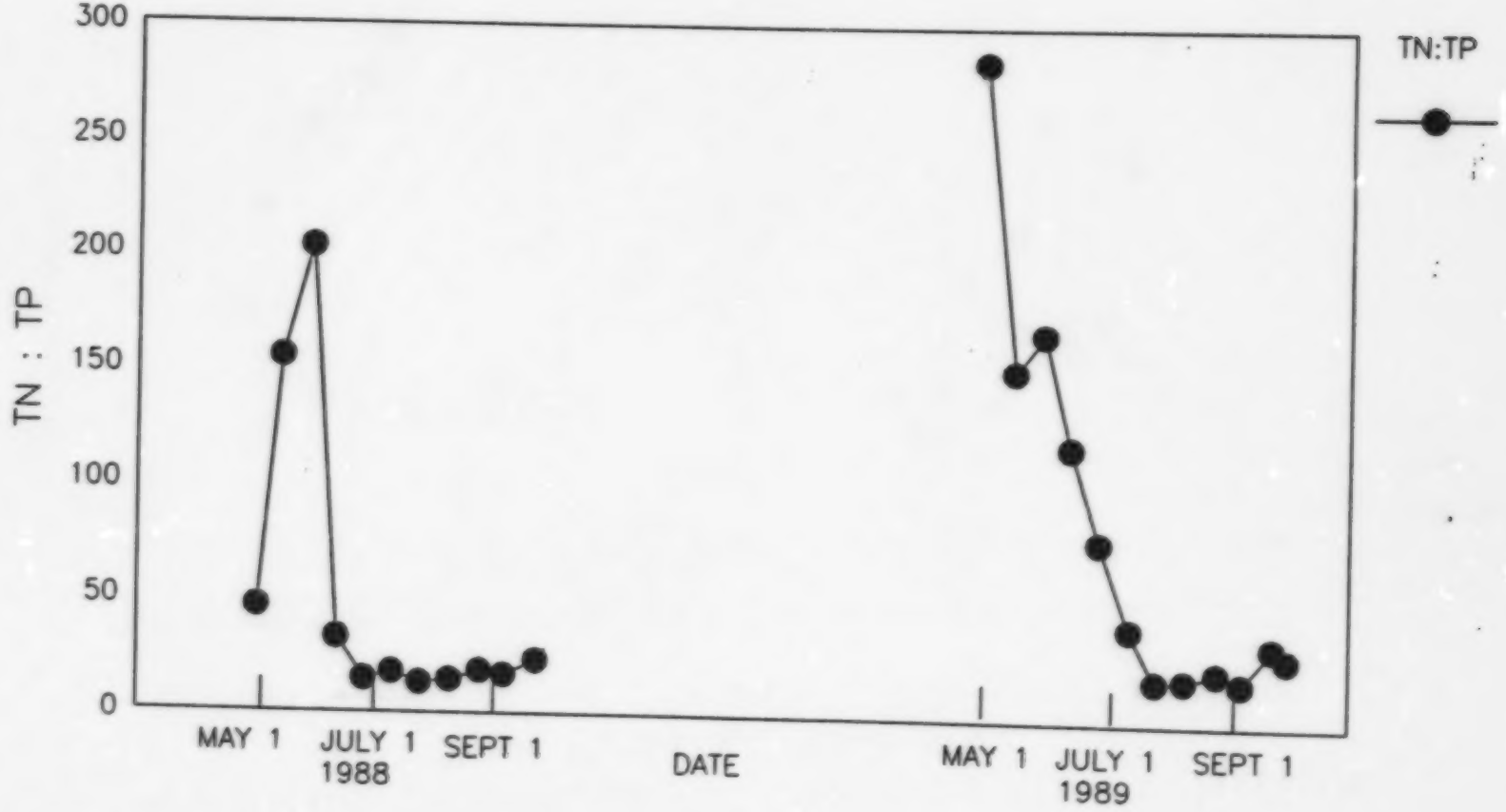


Fig. 10

Fanshawe Reservoir

Total Nitrogen : Total Phosphorus Ratio

1988/89

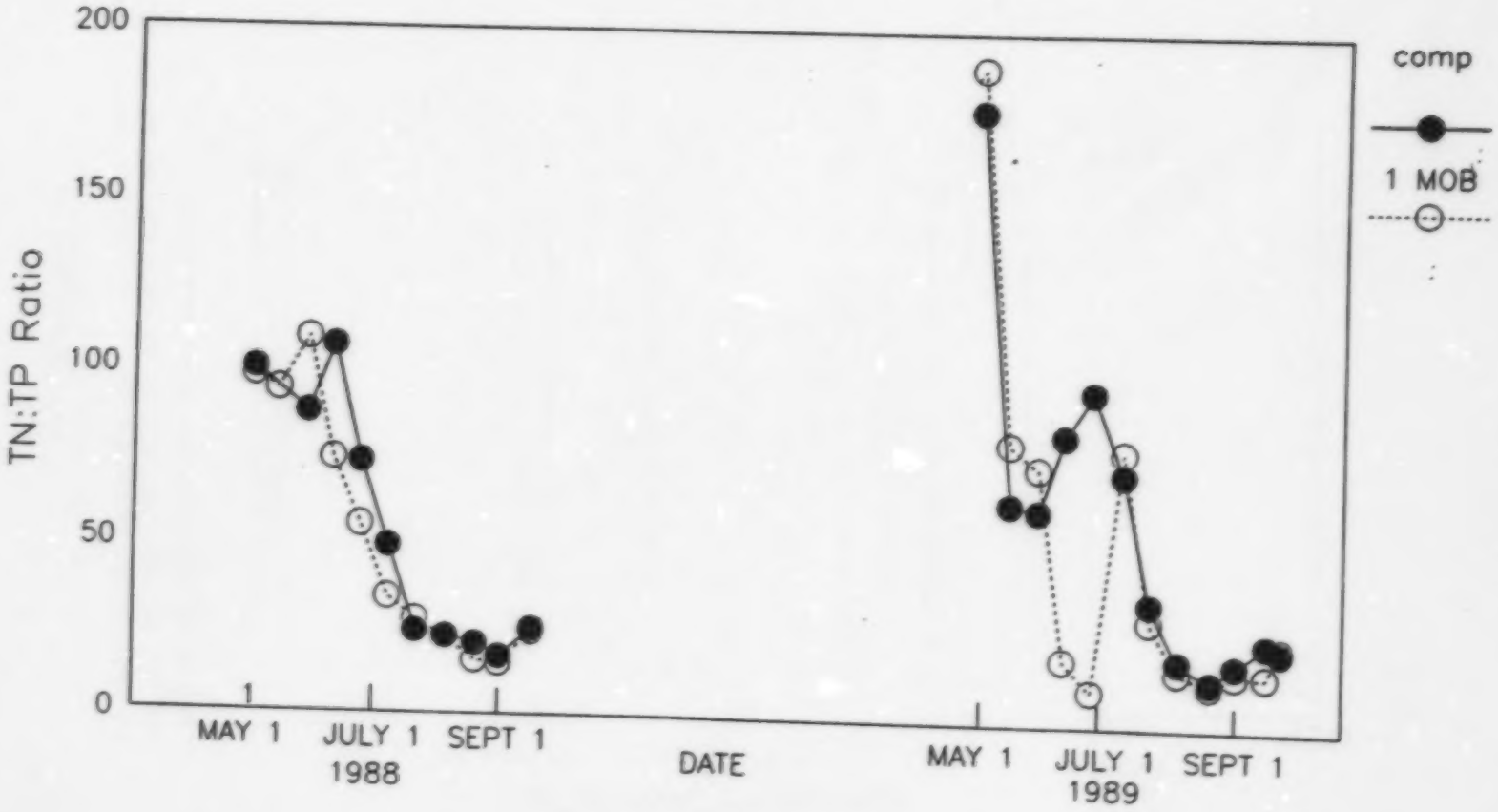


Fig. 11

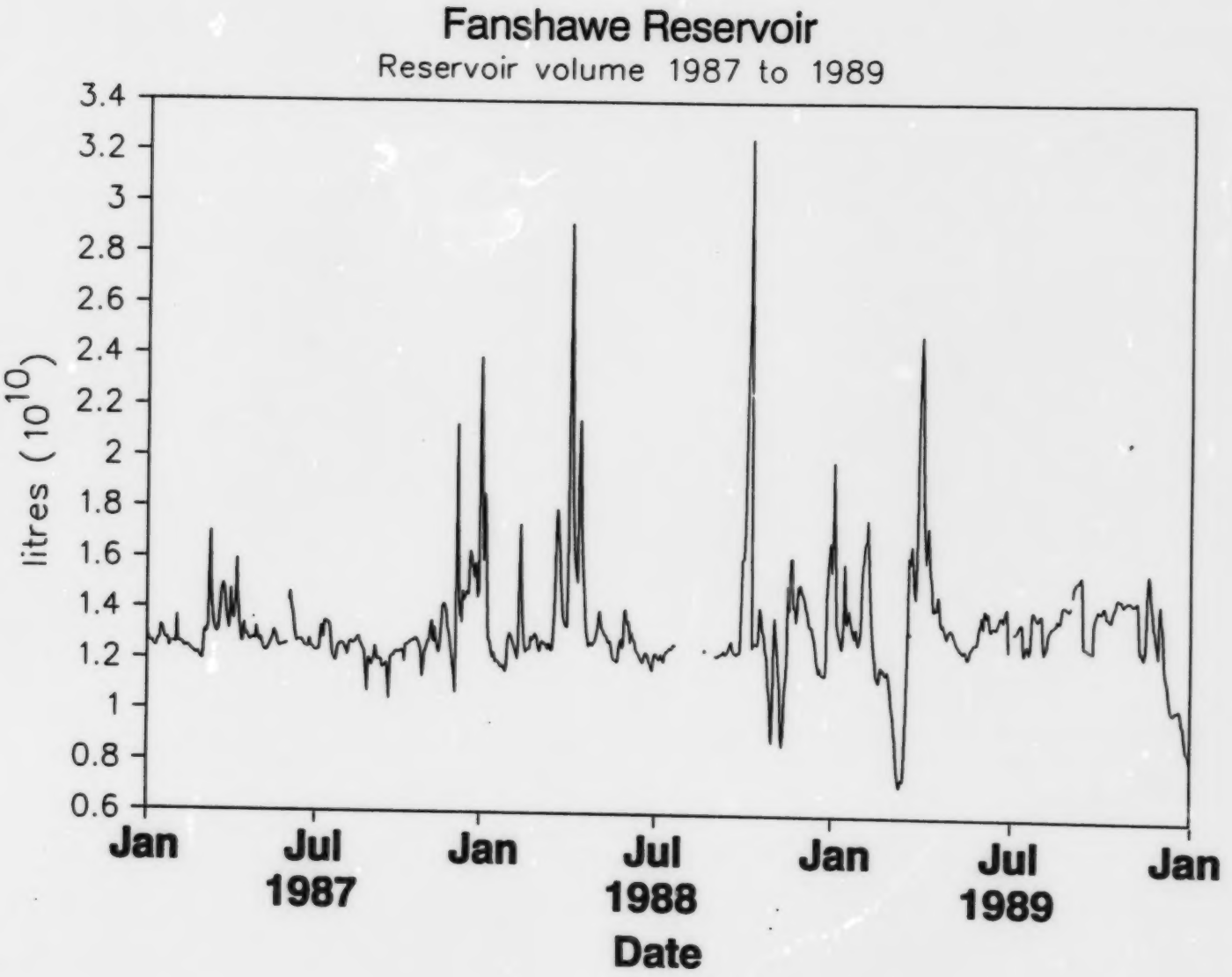

Fanshawe Reservoir
Reservoir volume 1987 to 1989
litres (10^{10})
3.4
3.2
3
2.8
2.6
2.4
2.2
2
1.8
1.6
1.4
1.2
1
0.8
0.6
Jan
Jul
1987
Jan
Jul
1988
Jan
Jul
1989
Jan
Date

Fig. 12

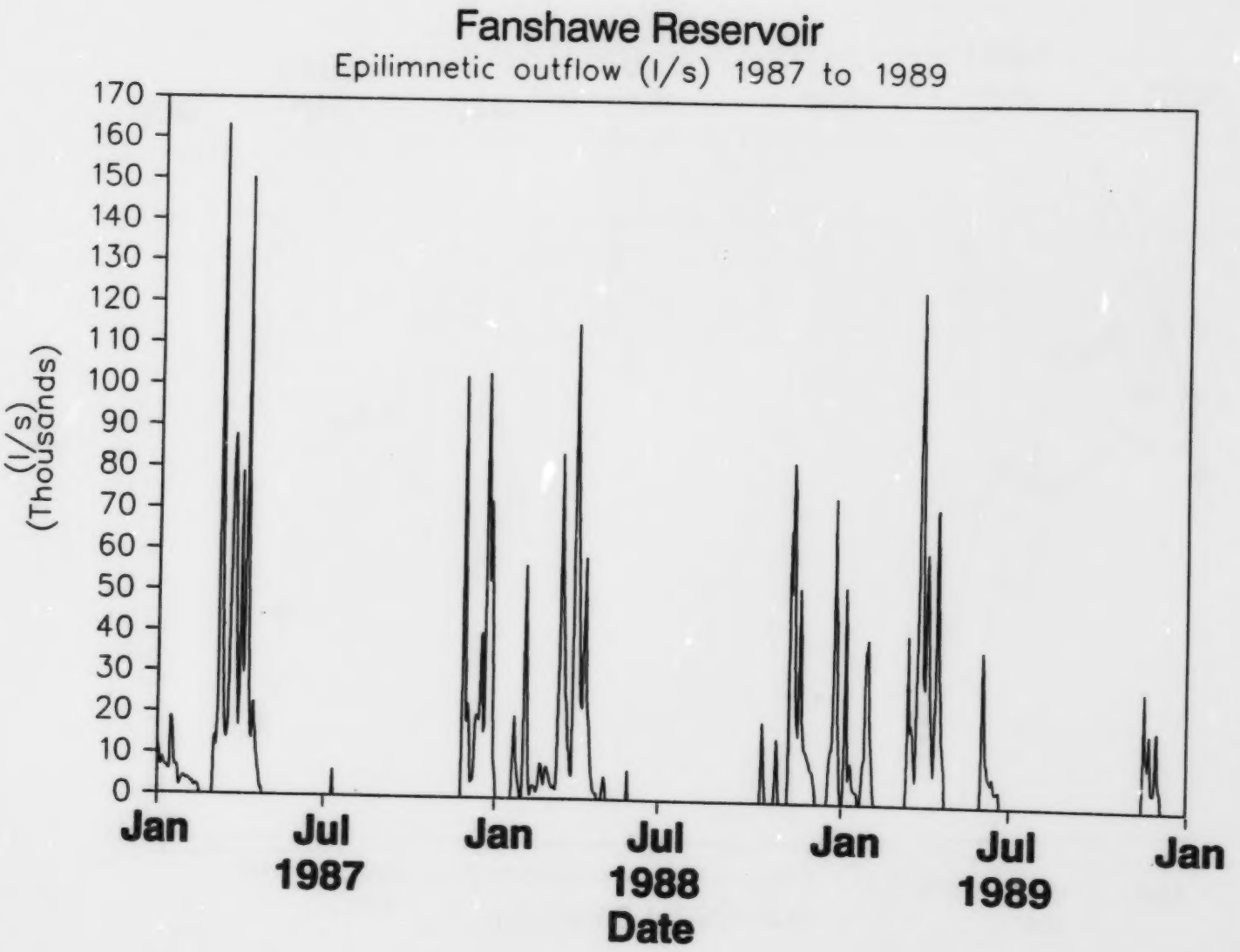

Fanshawe Reservoir
Epilimnetic outflow (l/s) 1987 to 1989
(l/s)
(Thousands)
170
160
150
140
130
120
110
100
90
80
70
60
50
40
30
20
10
0
Jan
Jul
1987
Jan
Jul
1988
Jan
Jul
1989
Jan
Date

Fig. 13

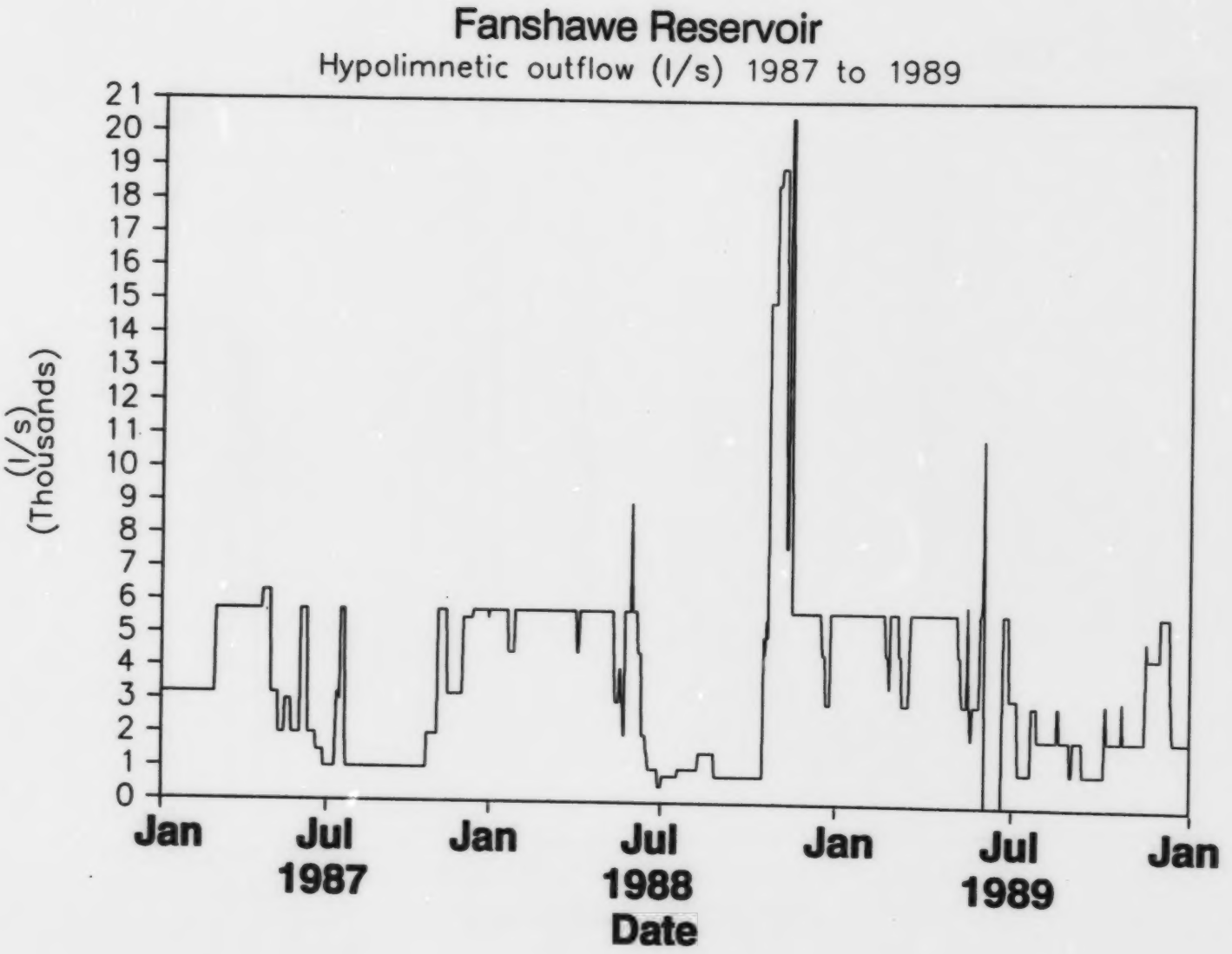

Fanshawe Reservoir
Hypolimnetic outflow (l/s) 1987 to 1989
(l/s)
(Thousands)
21
20
19
18
17
16
15
14
13
12
11
10
9
8
7
6
5
4
3
2
1
0
Jan
Jul
1987
Jan
Jul
1988
Jan
Jul
1989
Jan
Date

Fig. 14

Fanshawe Reservoir

Temperature (C)

1988 / 89

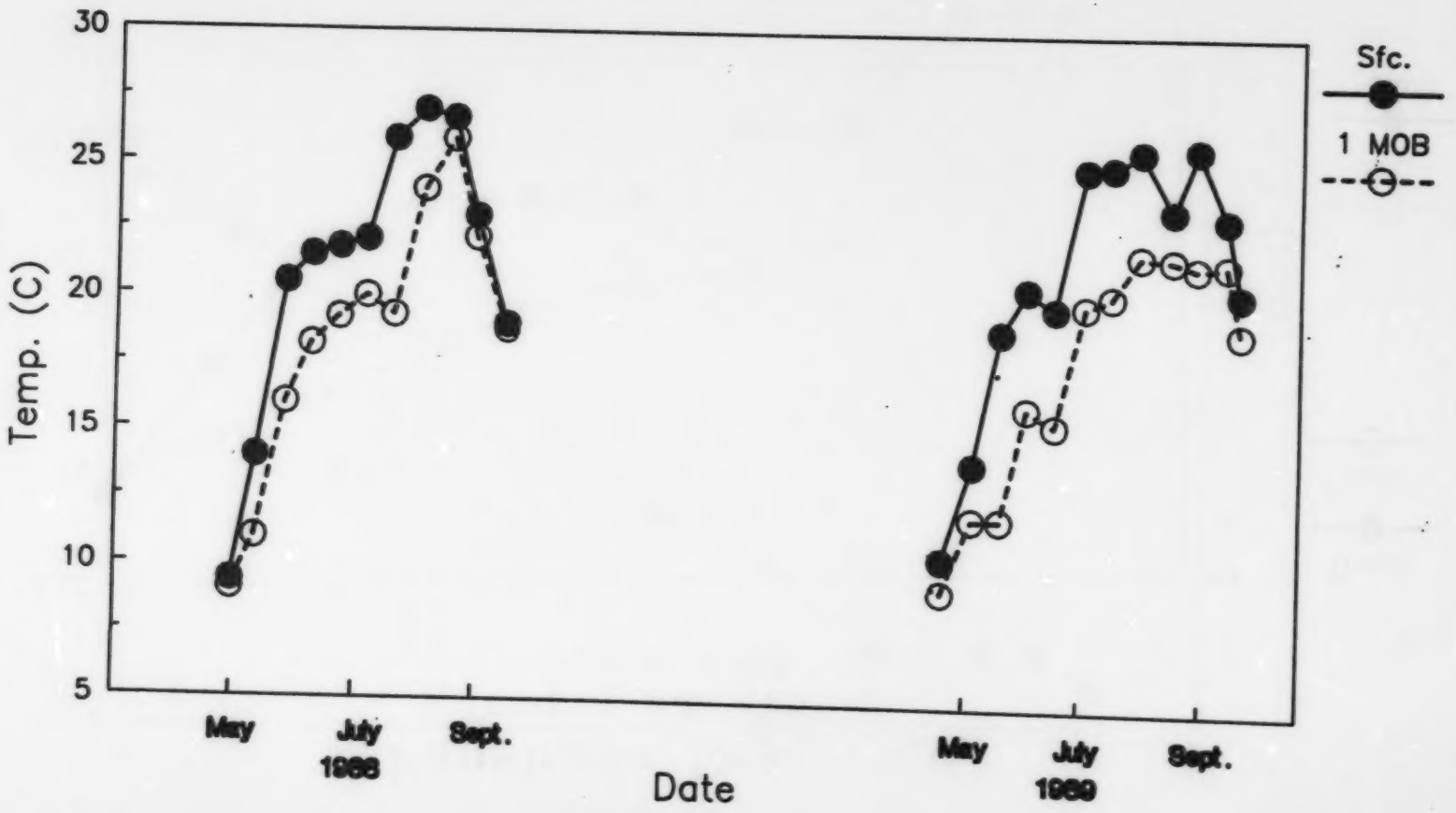


Fig. 15

Fanshawe Reservoir

Temp. / D.O. profile

July 4, 1988

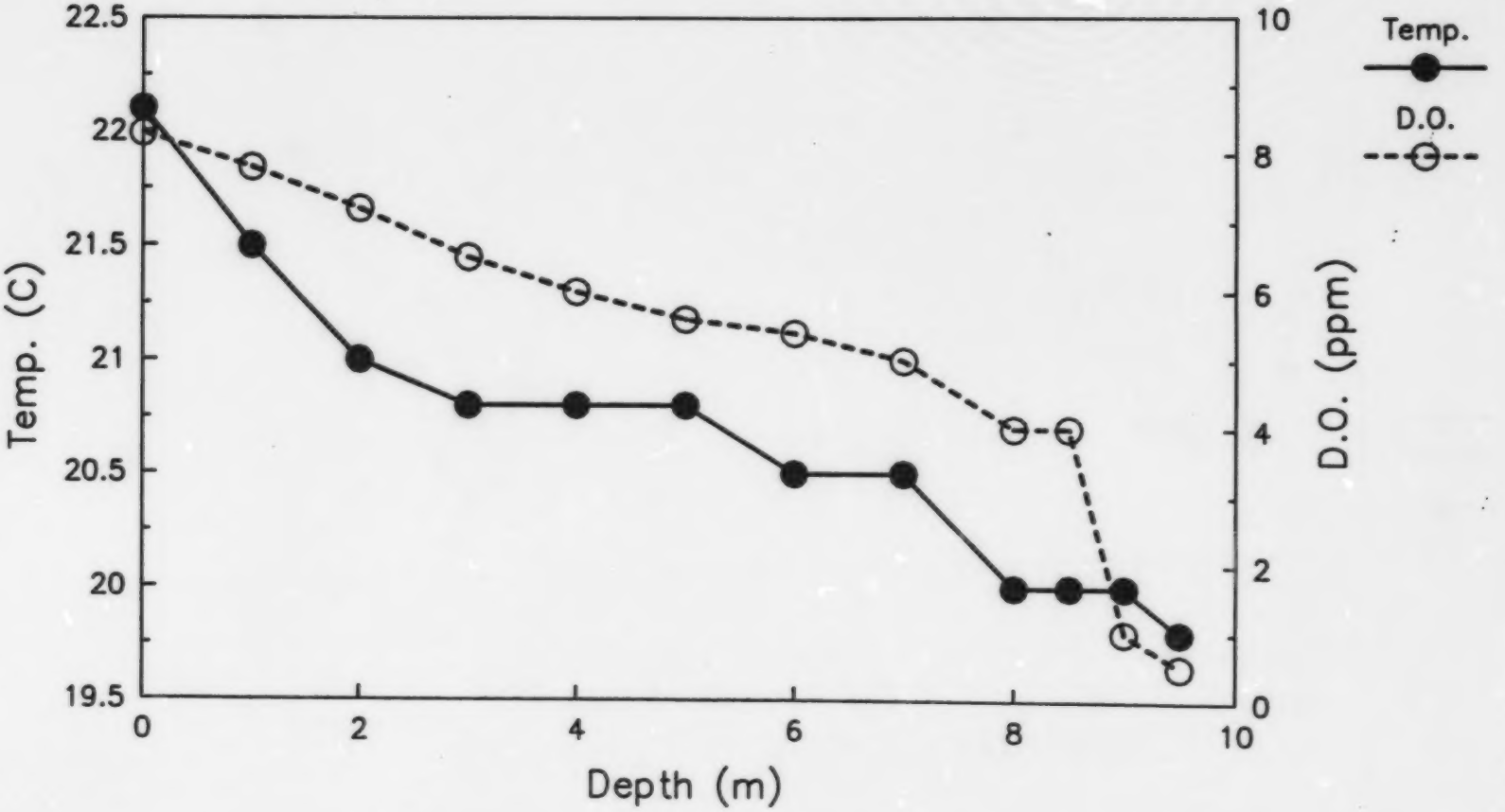


Fig. 16

Fanshawe Reservoir

Temp. / D.O. profile

Aug. 2, 1988

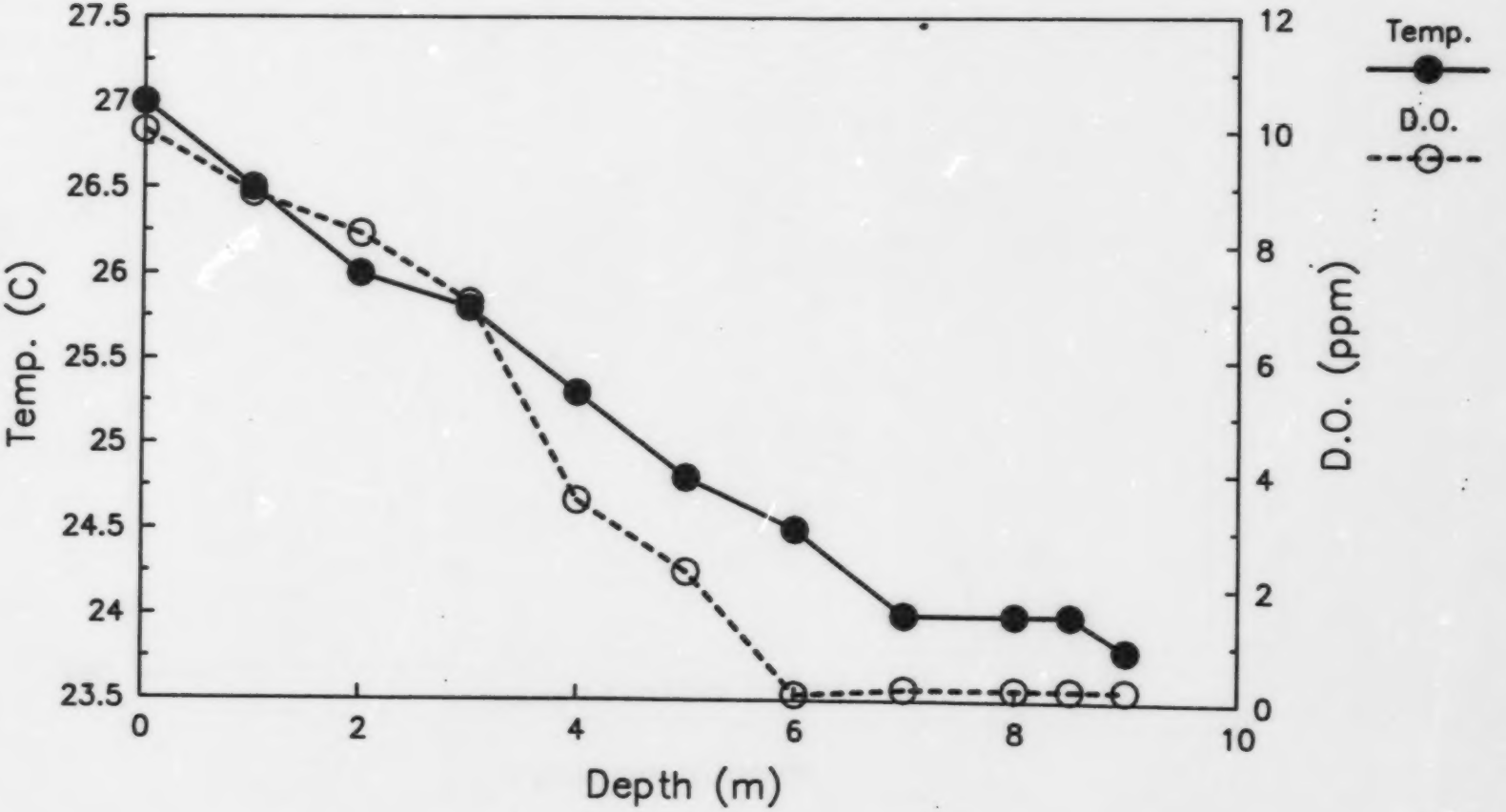


Fig. 17

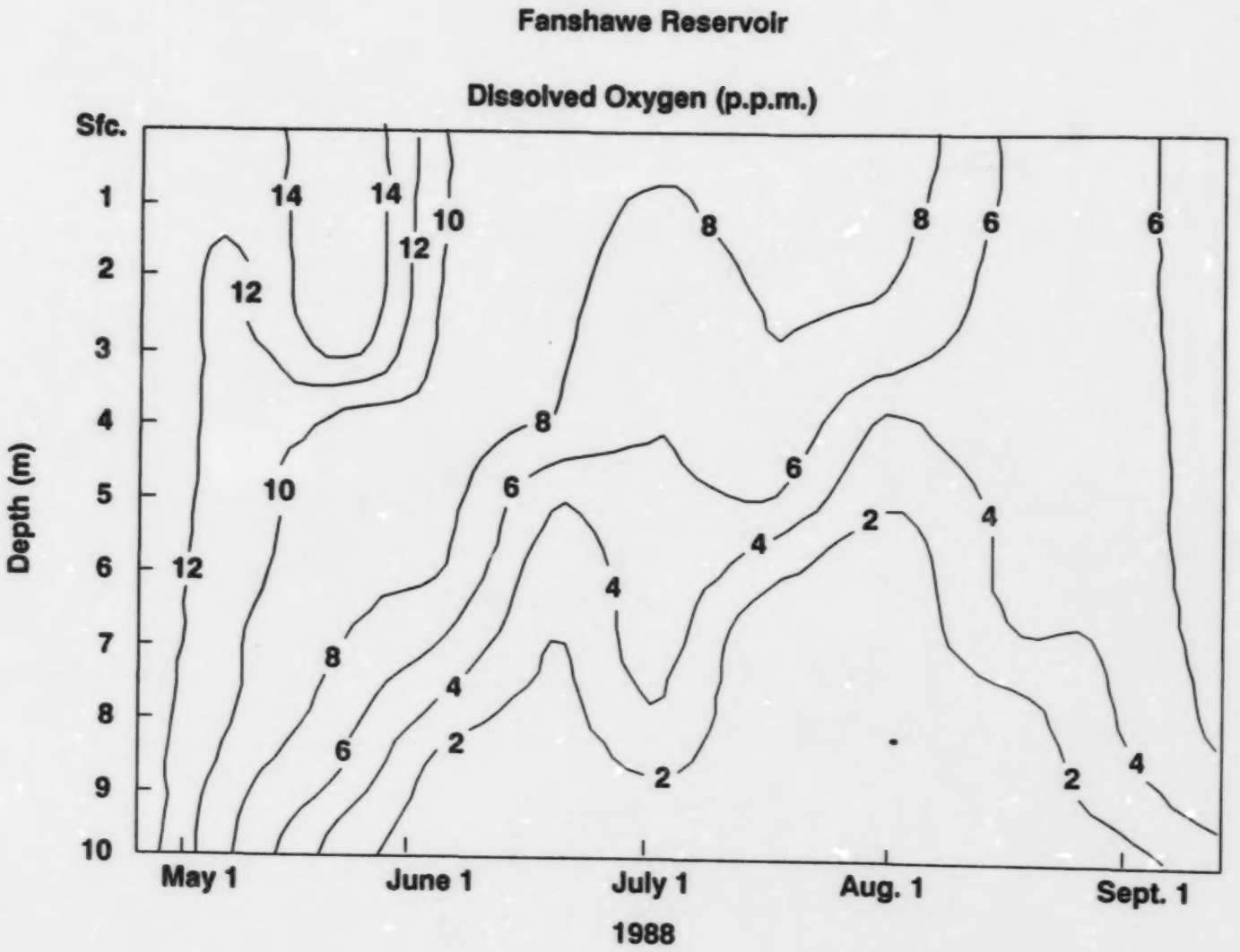

Fanshawe Reservoir
Dissolved Oxygen (p.p.m.)
Sfc.
1
2
3
4
5
6
7
8
9
10
Depth (m)
May 1
June 1
July 1
Aug. 1
Sept. 1
1988
14
14
12
10
12
12
10
8
8
8
6
6
6
6
4
4
4
4
2
2
2
2
4
8
6
6
4
2

Fig. 18

Fanshawe Reservoir

Dissolved Oxygen (p.p.m.)

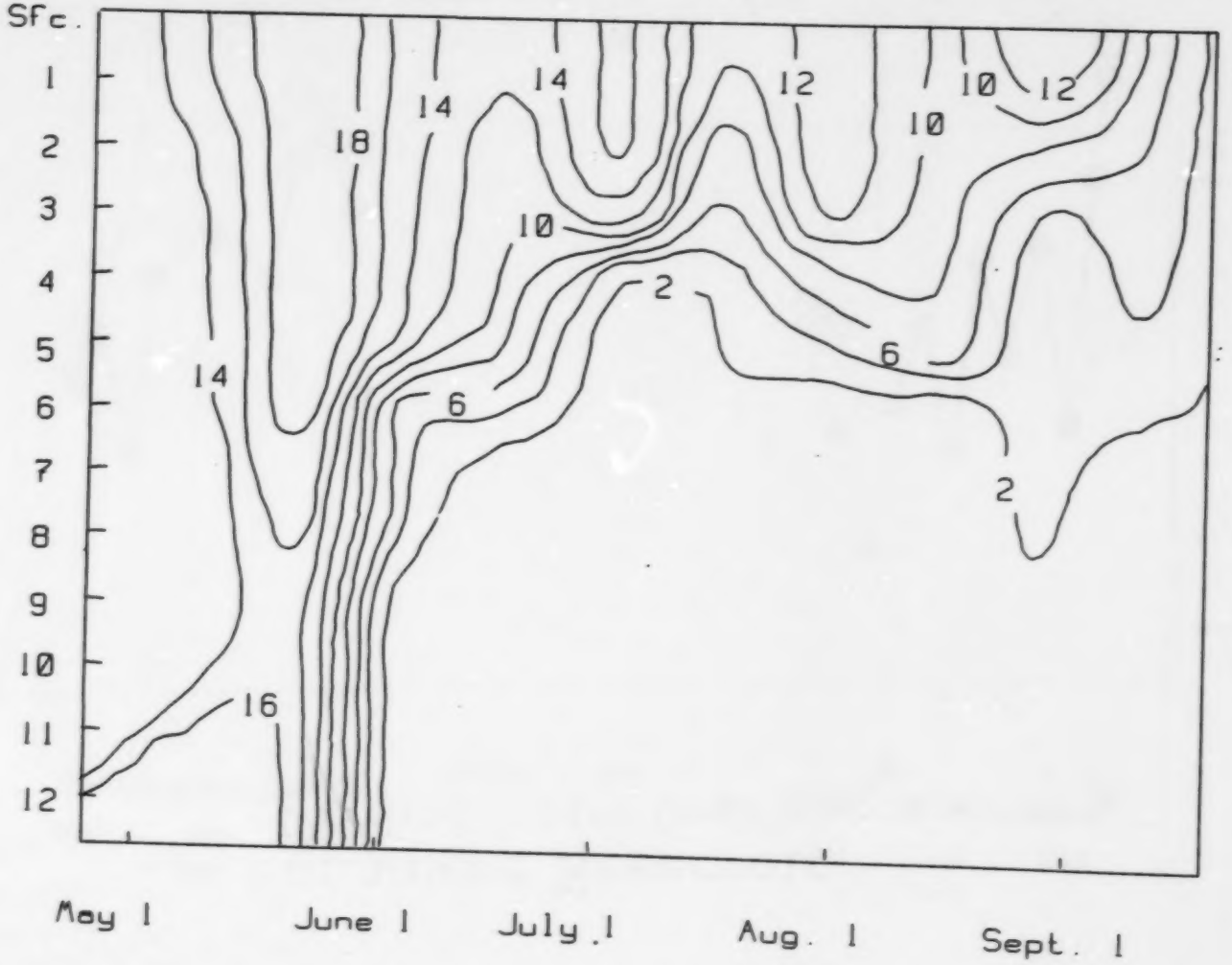


Fig. 19

Fanshawe Reservoir

Dissolved Oxygen (mg/l)

1988 / 89

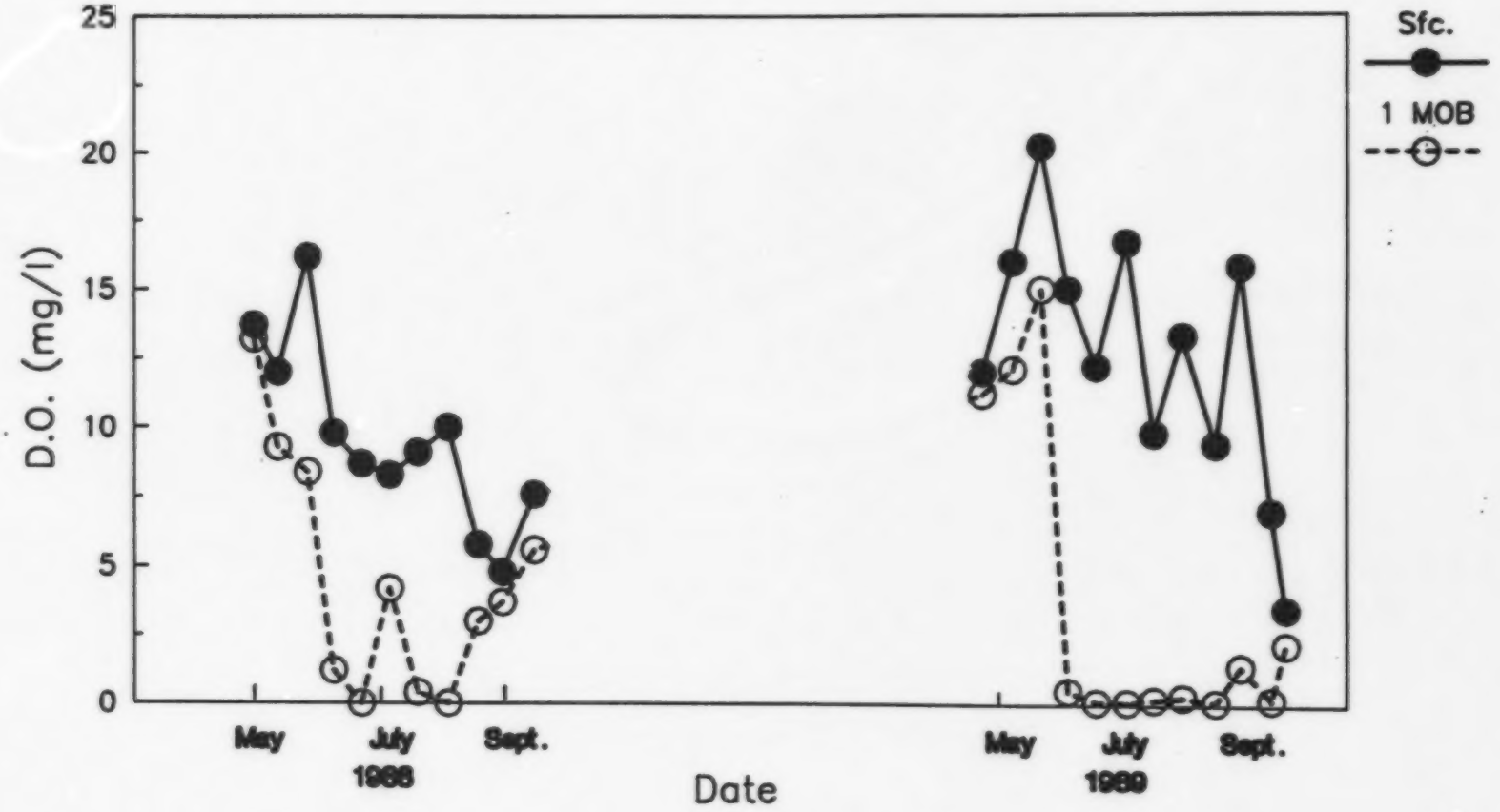


Fig. 20

Fanshawe Reservoir

Manganese (mg/l as Mn)

1988 / 89

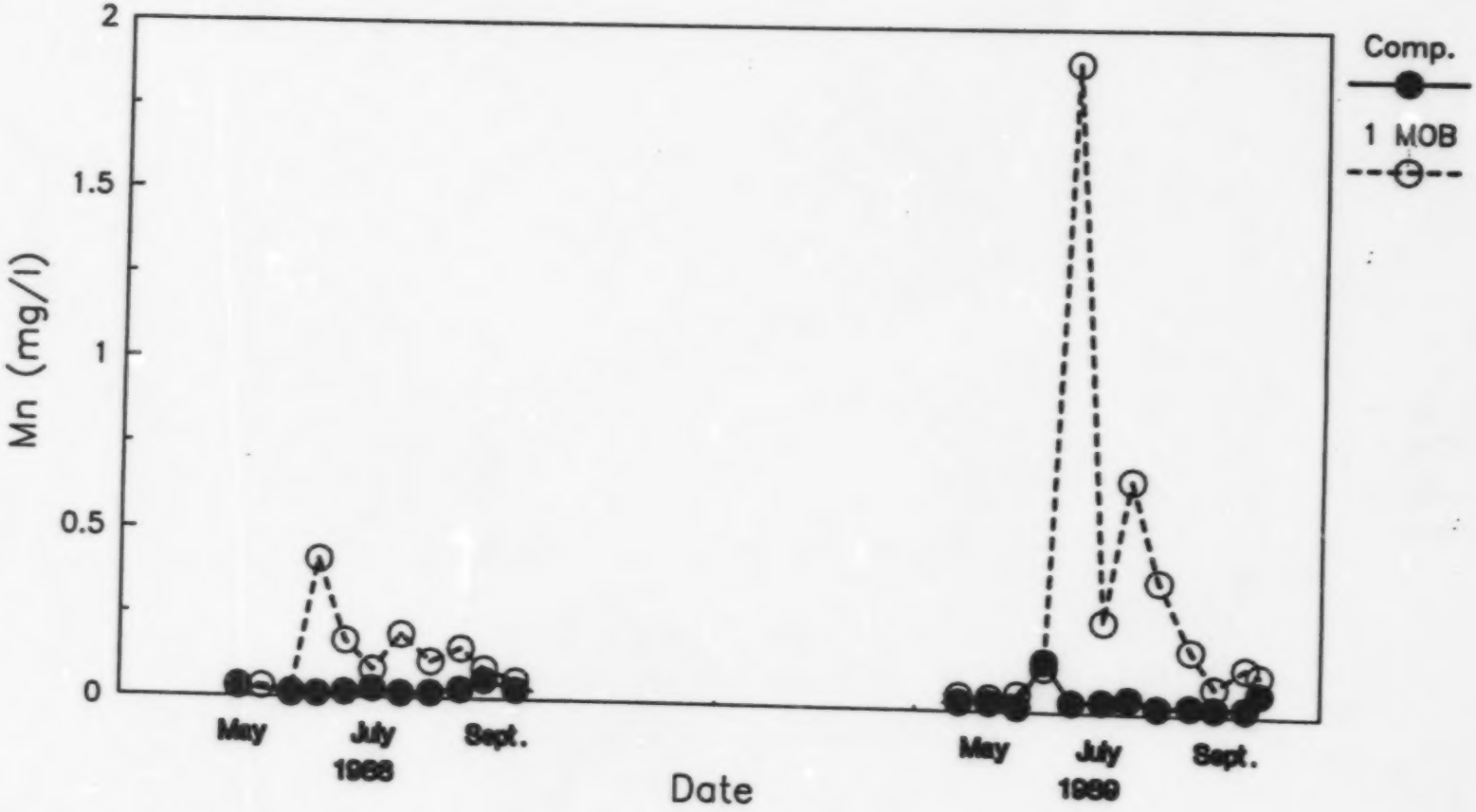


Fig. 21

Fanshawe Reservoir
Iron (mg/l as Fe)
1988 / 89

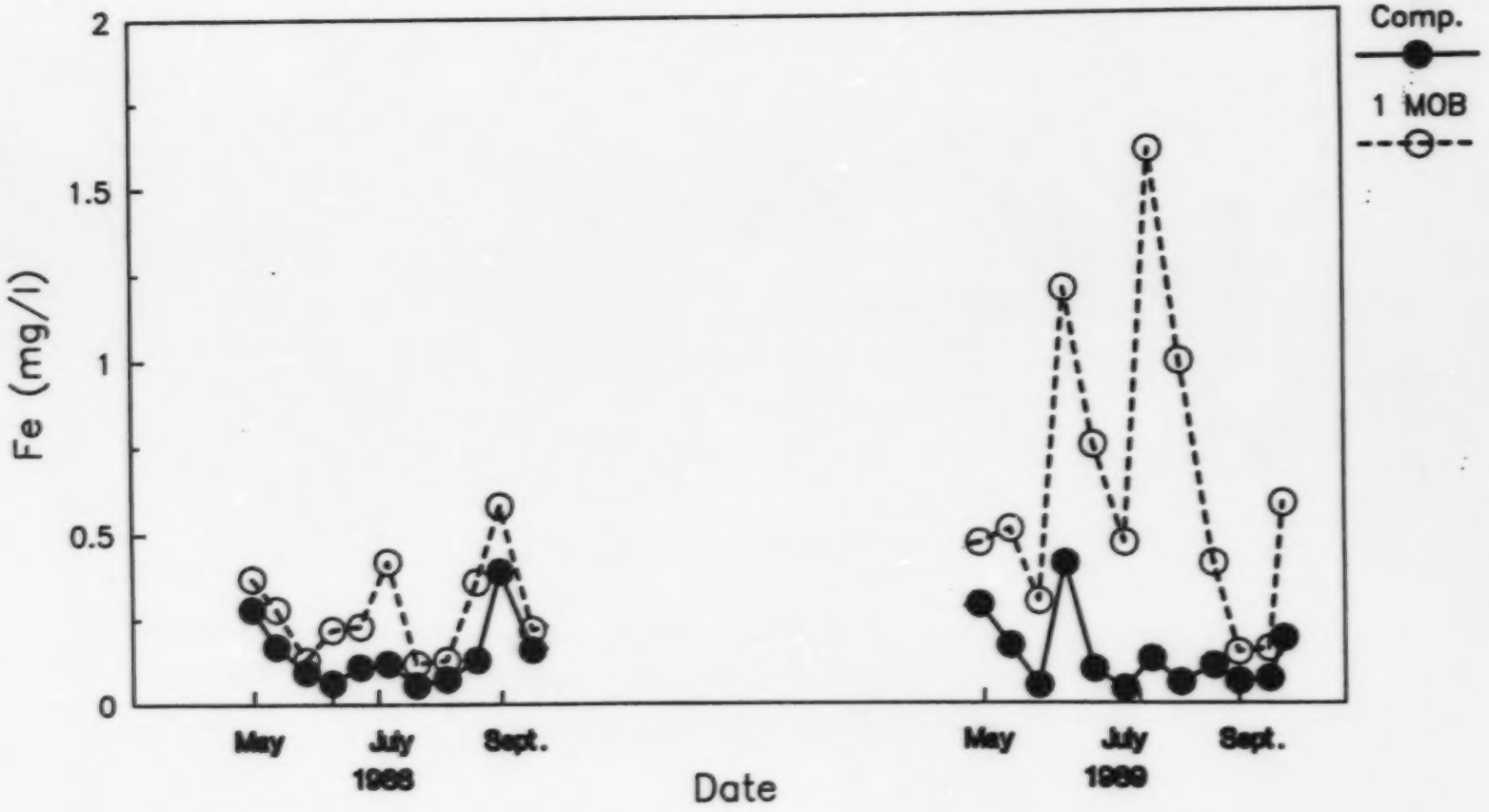


Fig. 22

Fanshawe Reservoir

Orthophosphorus (mg/l as P)

1988 / 89

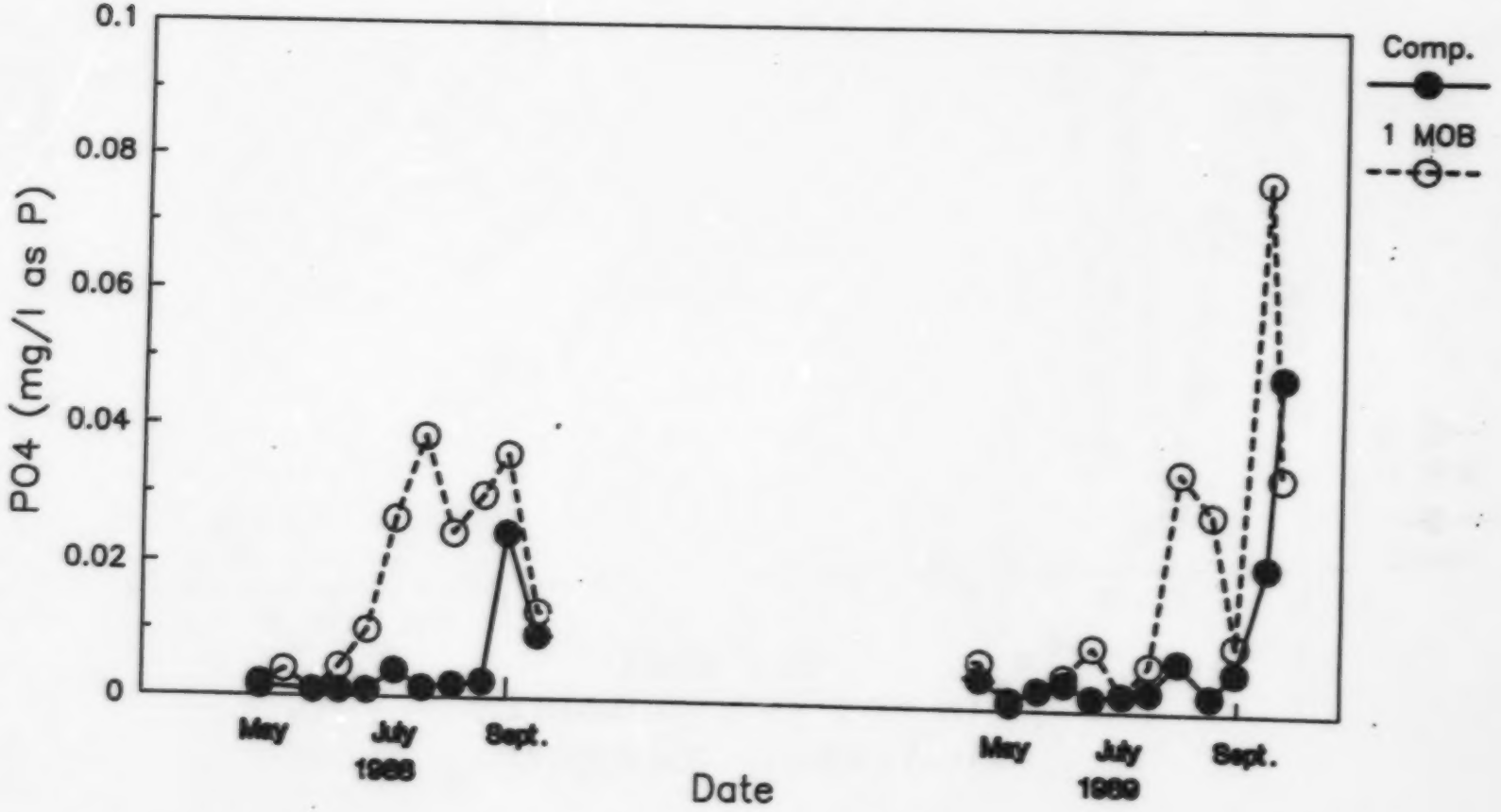


Fig. 23

Fanshawe Reservoir

Ammonium (mg/l as N)

1988 / 89

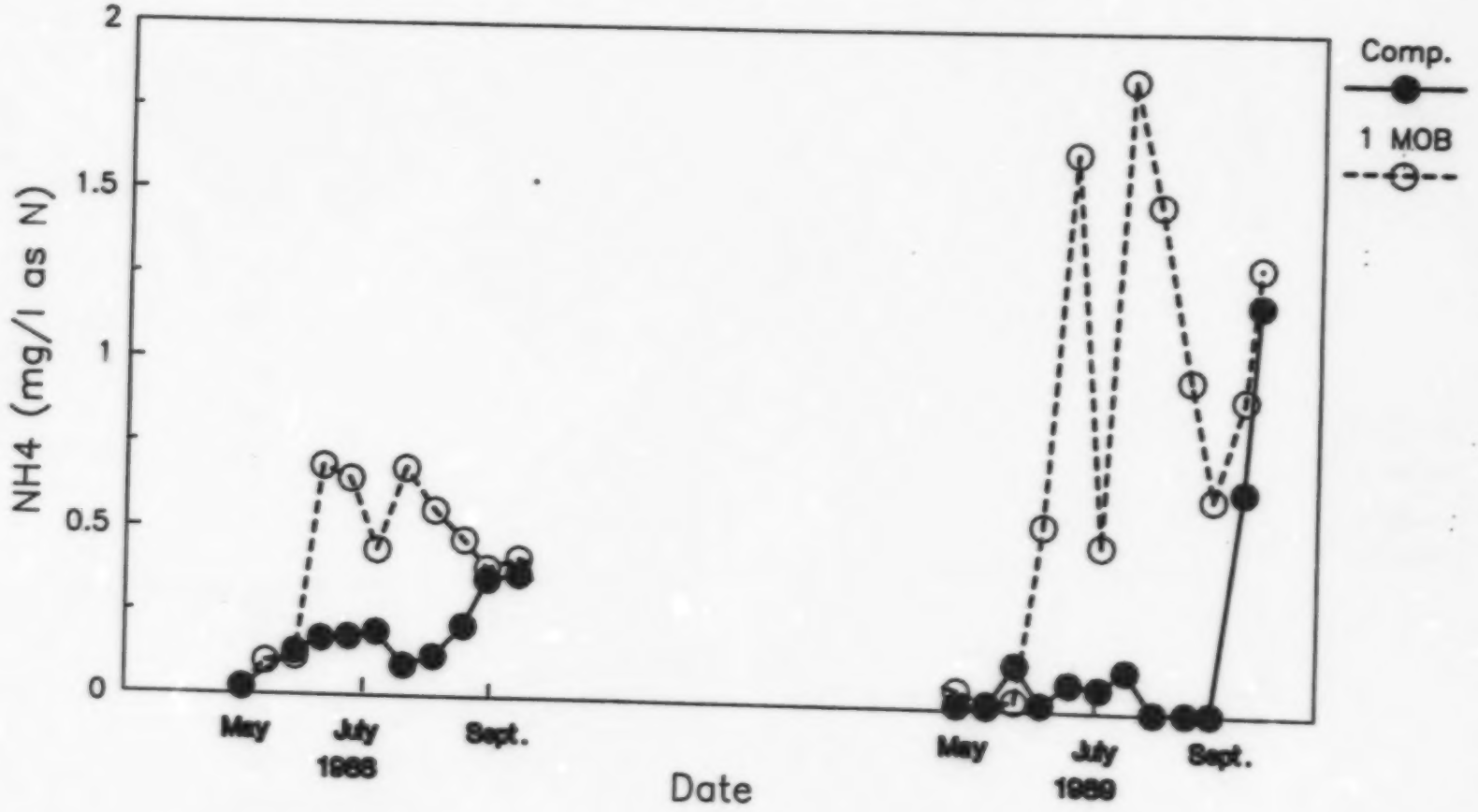


Fig. 24

Fanshawe Reservoir
Turbidity (FTU)
1988/89

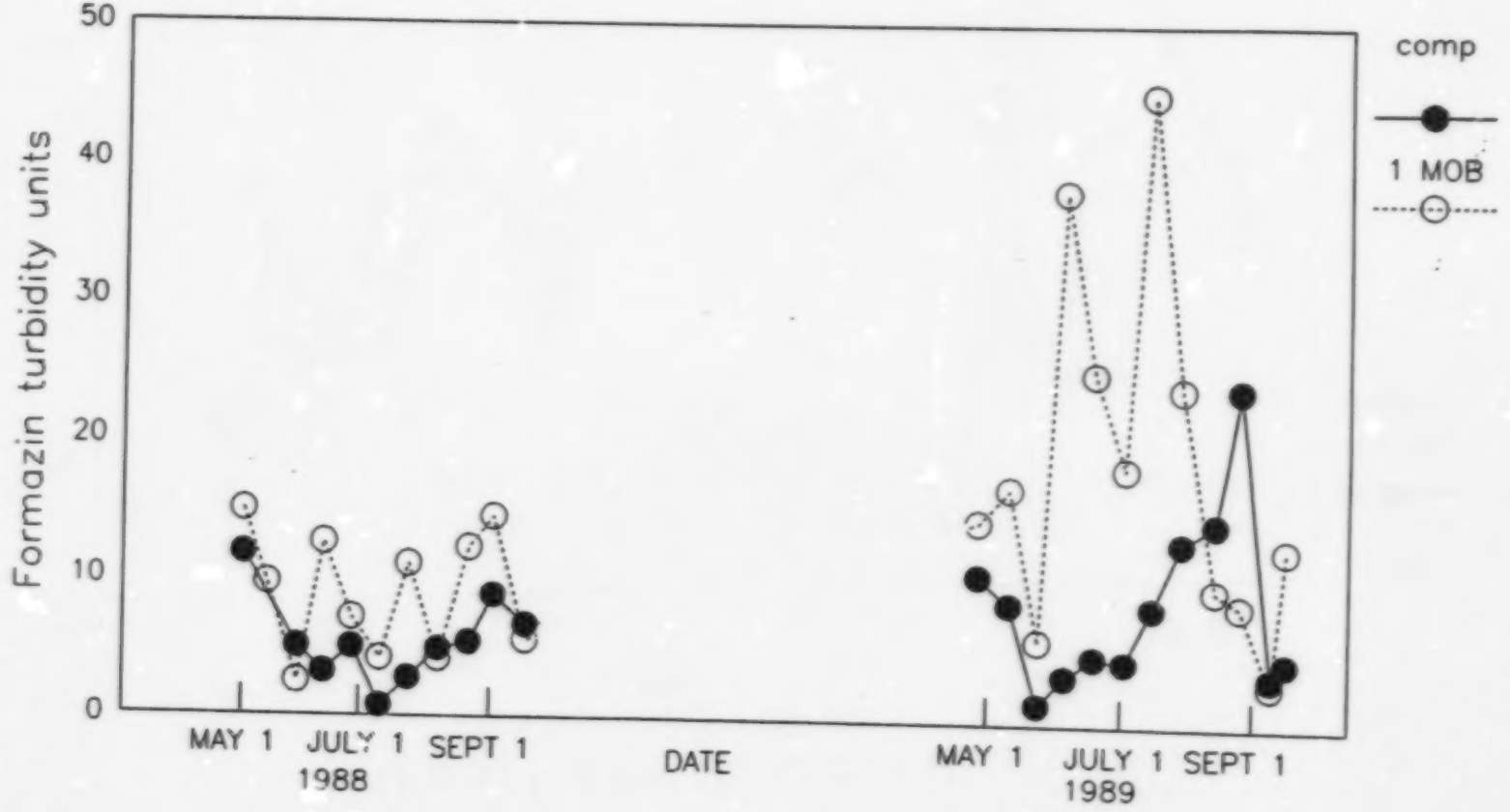


Fig. 25

Fanshawe Reservoir

Conductivity (µS/cm)

1988/89

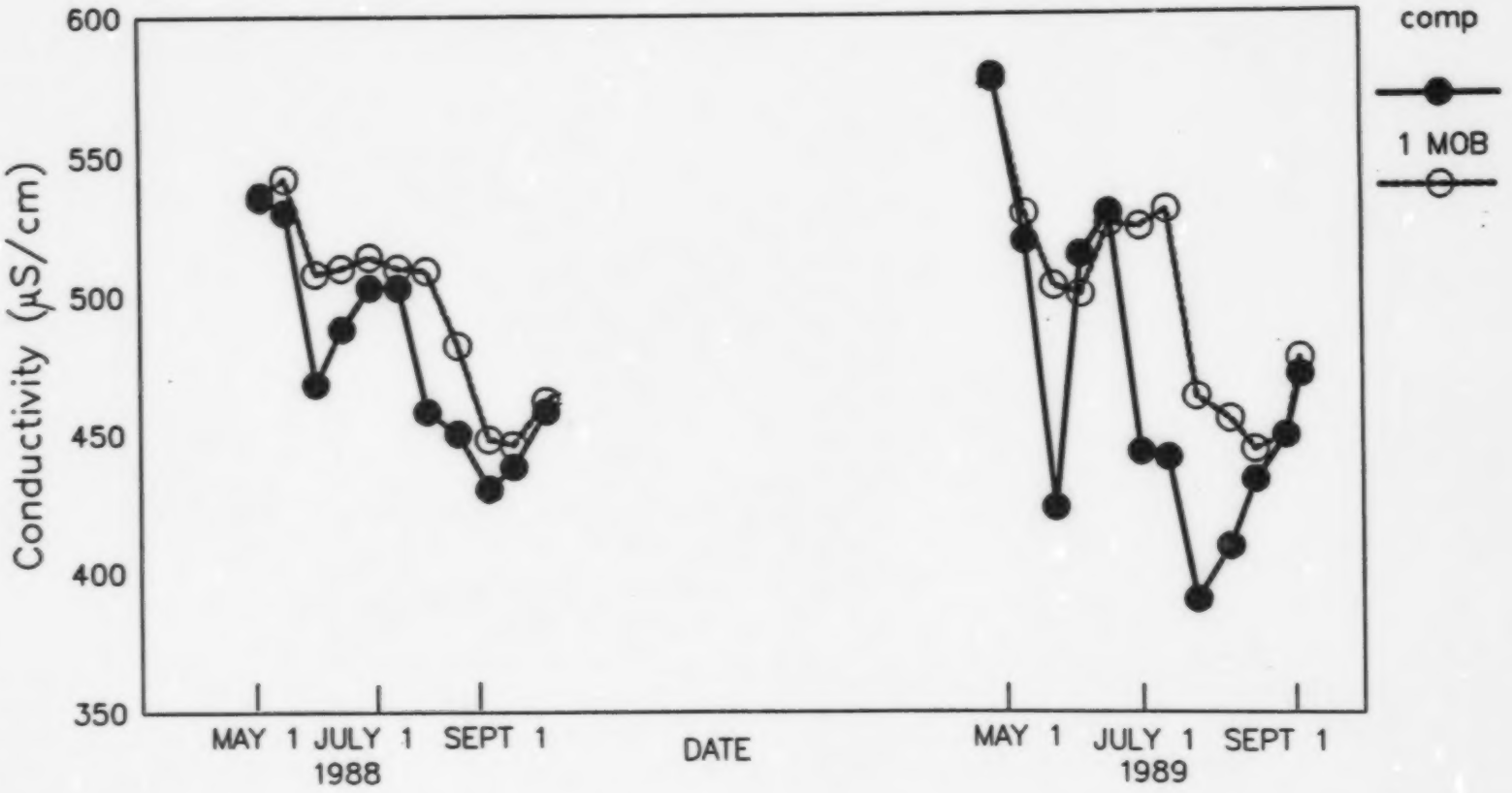


Fig. 26

Fanshawe Reservoir

Dissolved Inorganic Carbon (mg/l)

1988/89

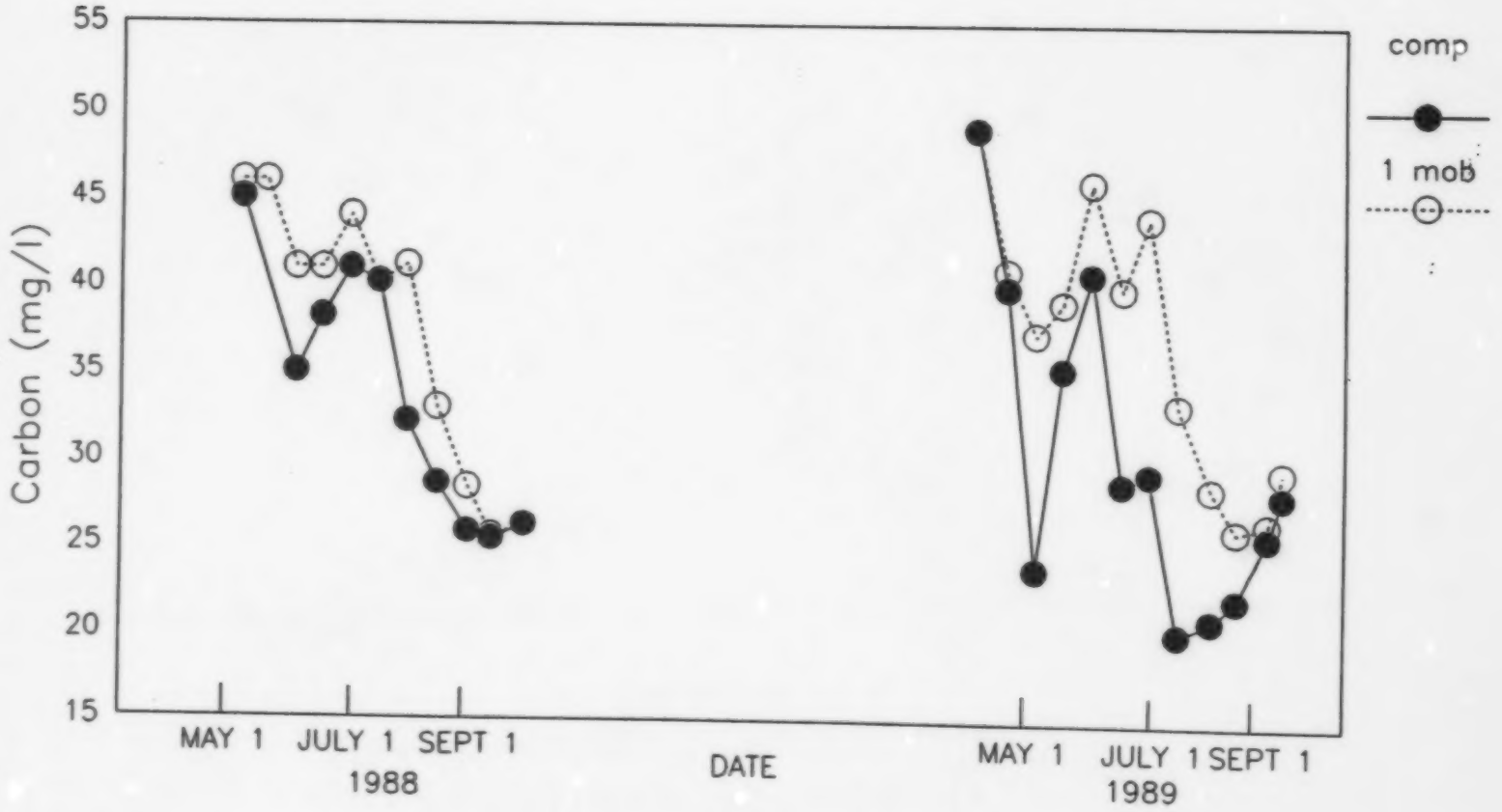


Fig. 27

Fanshawe Reservoir

Dissolved Organic Carbon (mg/l)

1988/89

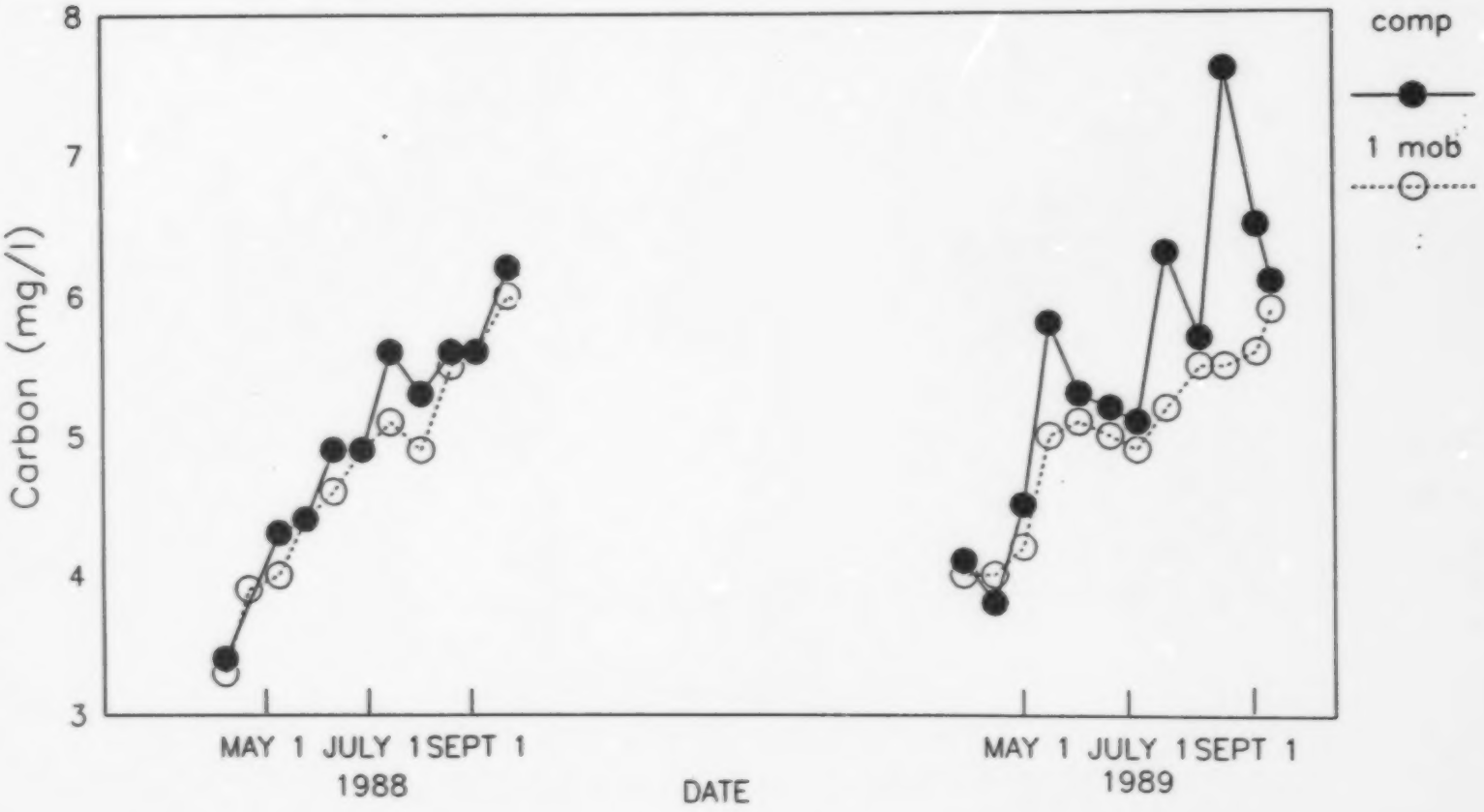


Fig. 28

Fanshawe Reservoir
Total Phosphorus (mg/l)
1988/89

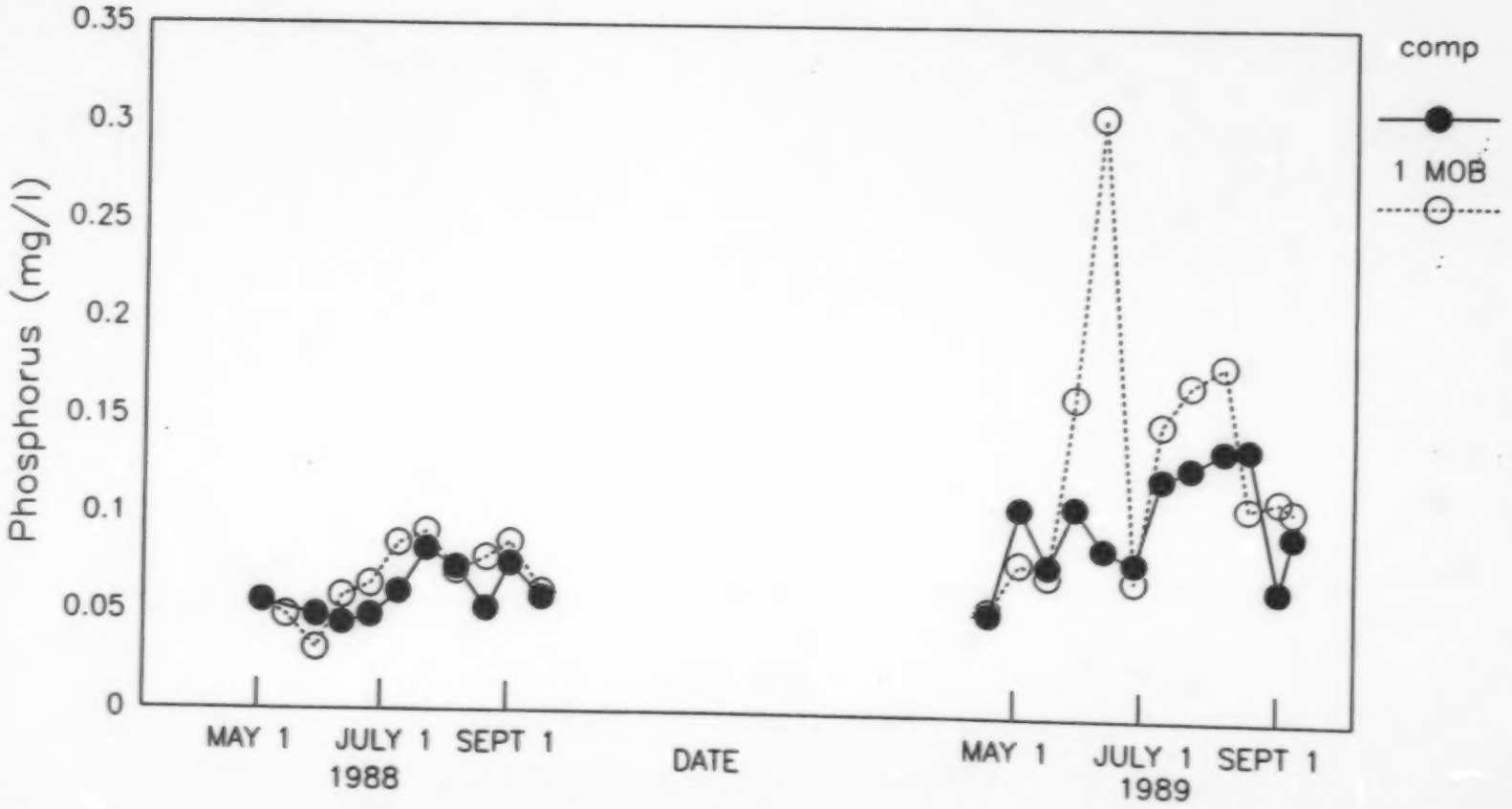


Fig. 29

Fanshawe Reservoir

Kjeldahl Nitrogen (mg/l)

1988/89

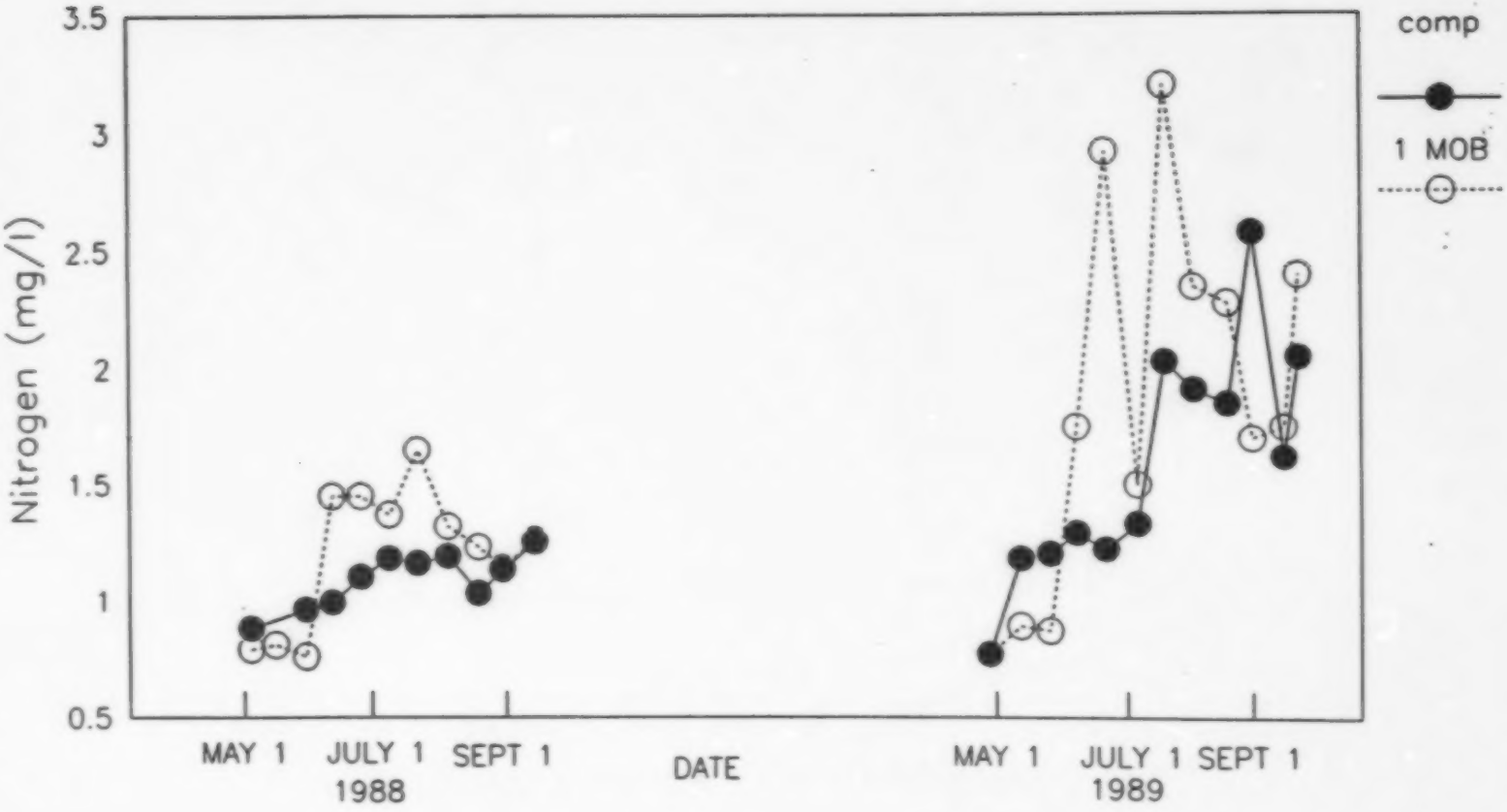


Fig. 30

Fanshawe Reservoir

Secchi Disc (M)

1988/89

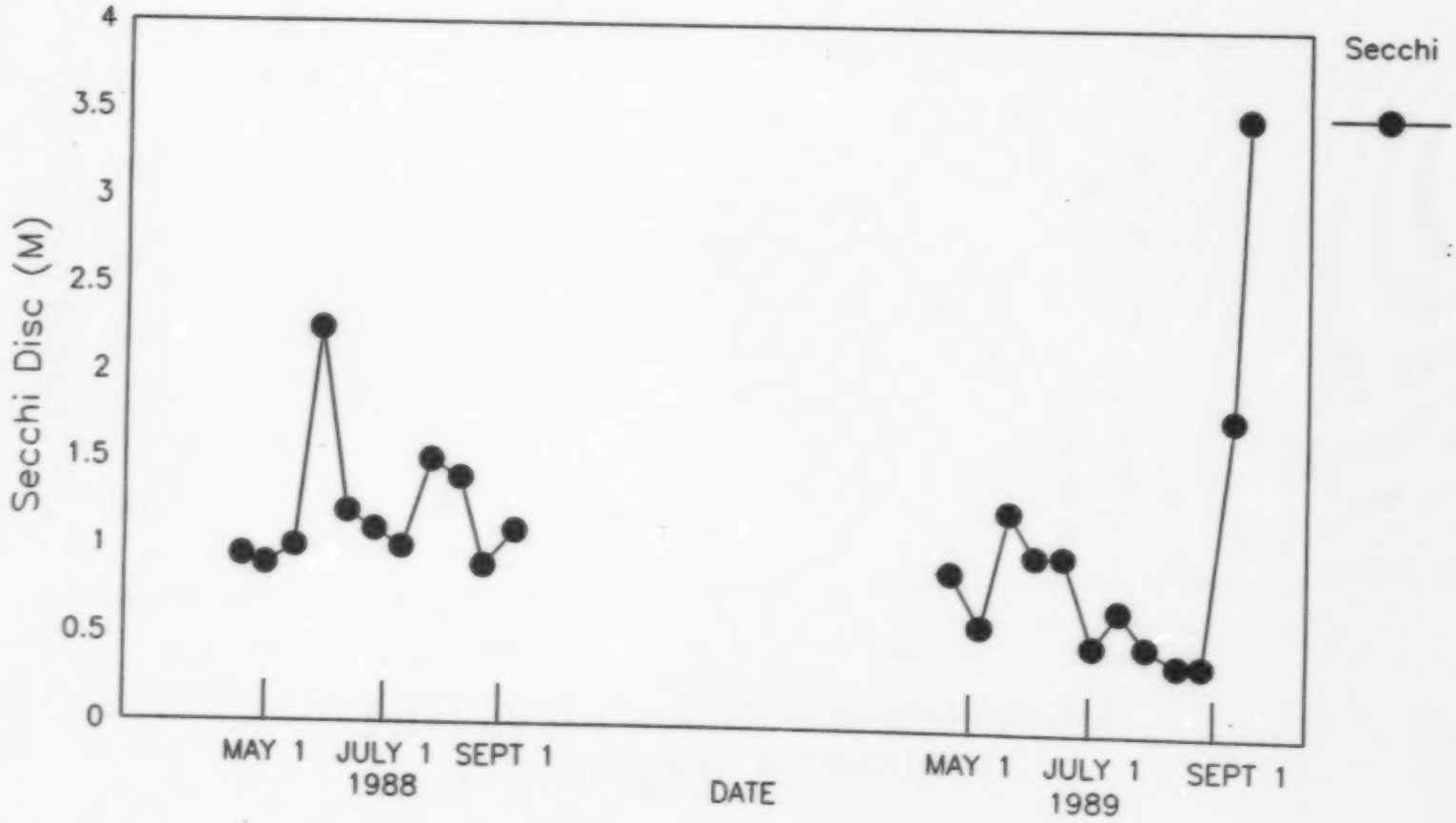


Fig. 31

Fanshawe Reservoir

Total Chlorophyll a / Chlorophyll b

1988/89

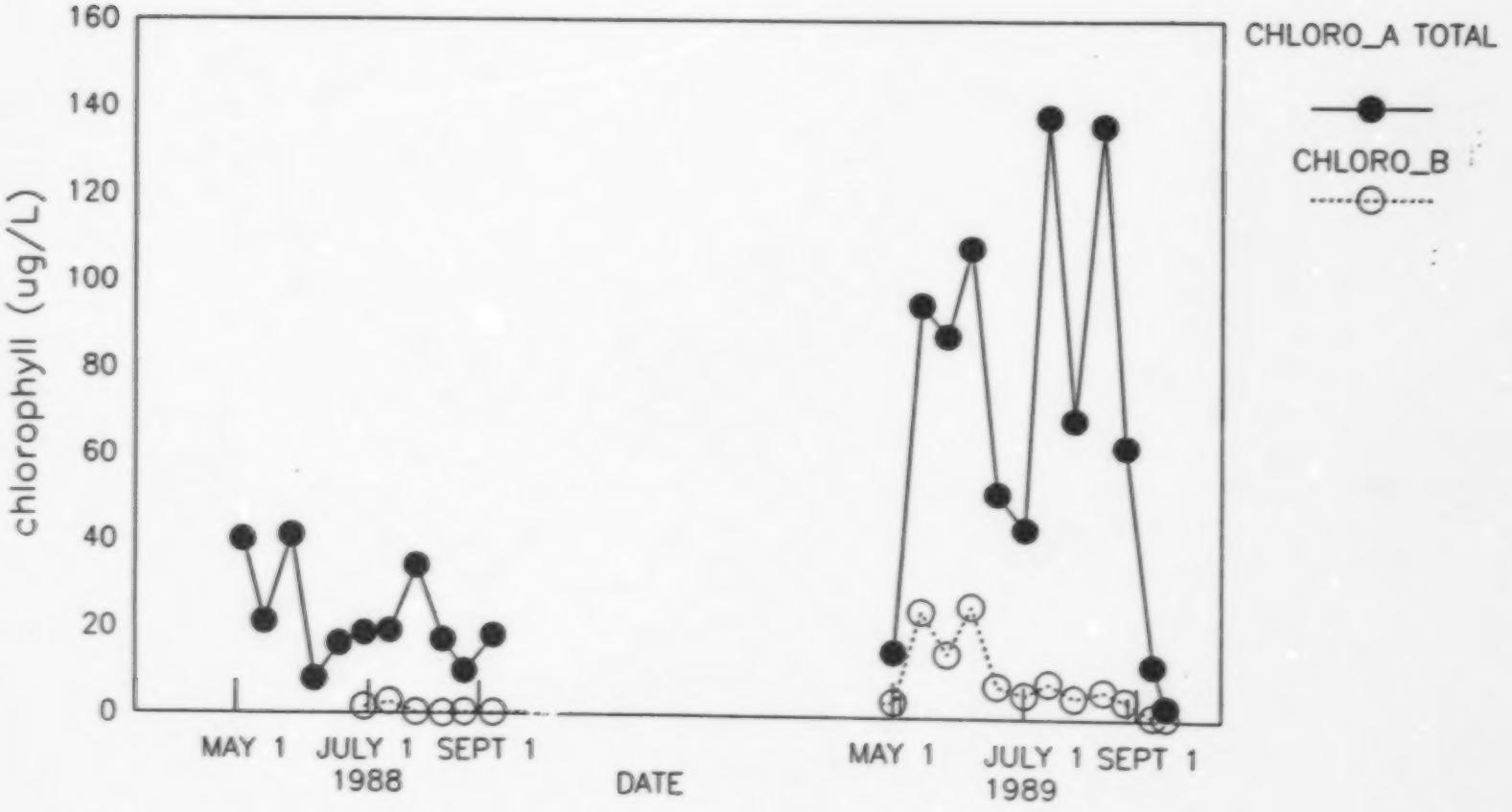


Fig. 32

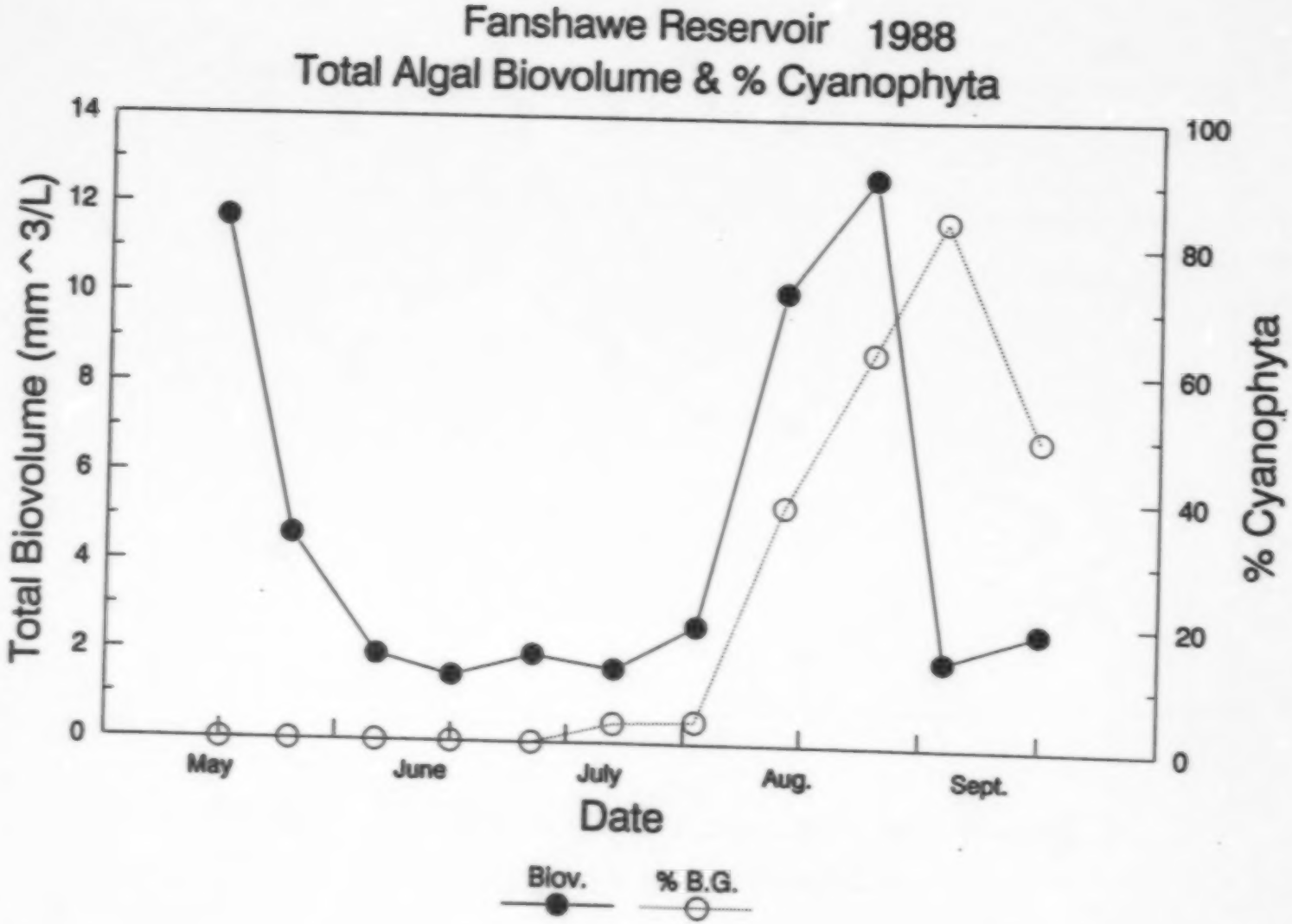

Fanshawe Reservoir 1988
Total Algal Biovolume & % Cyanophyta
Total Biovolume (mm^3/L)
% Cyanophyta
0
2
4
6
8
10
12
14
0
20
40
60
80
100
May
June
July
Aug.
Sept.
Date
Biov.
% B.G.

Fig. 33

Fanshawe Reservoir
Phytoplankton
1988

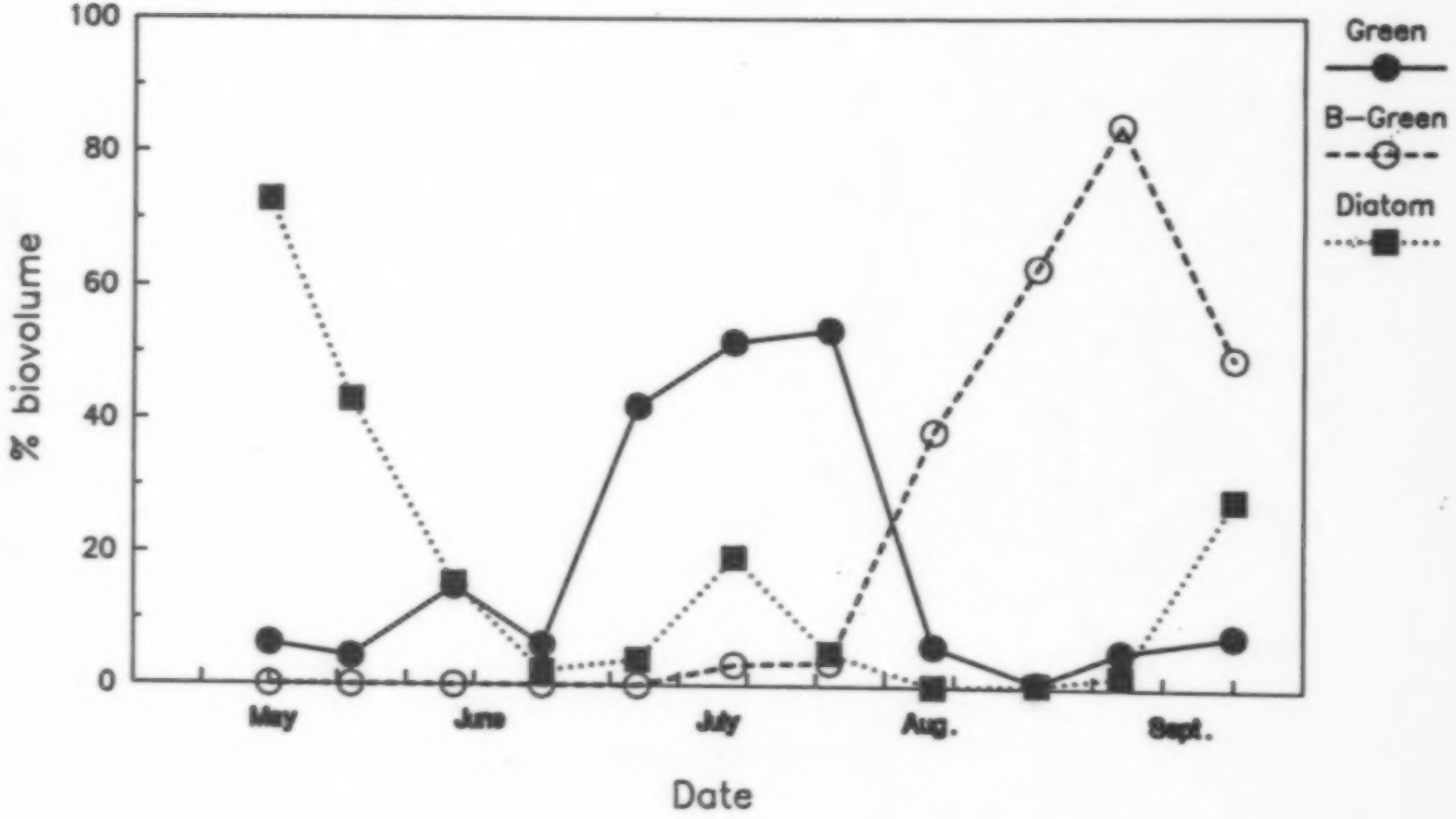


Fig. 34

Fanshawe Reservoir

Phytoplankton as a % of Total Biovolume

1988

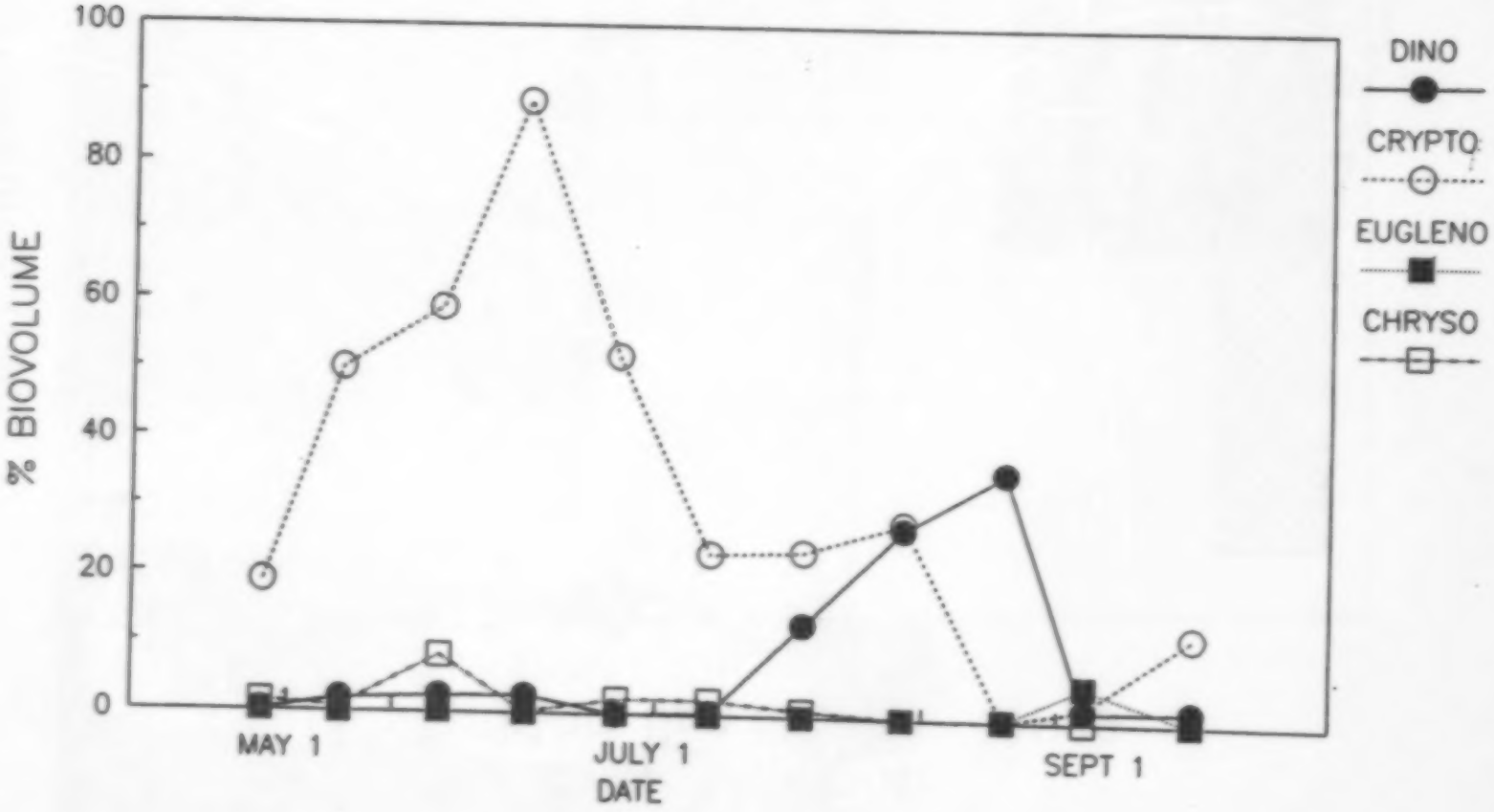
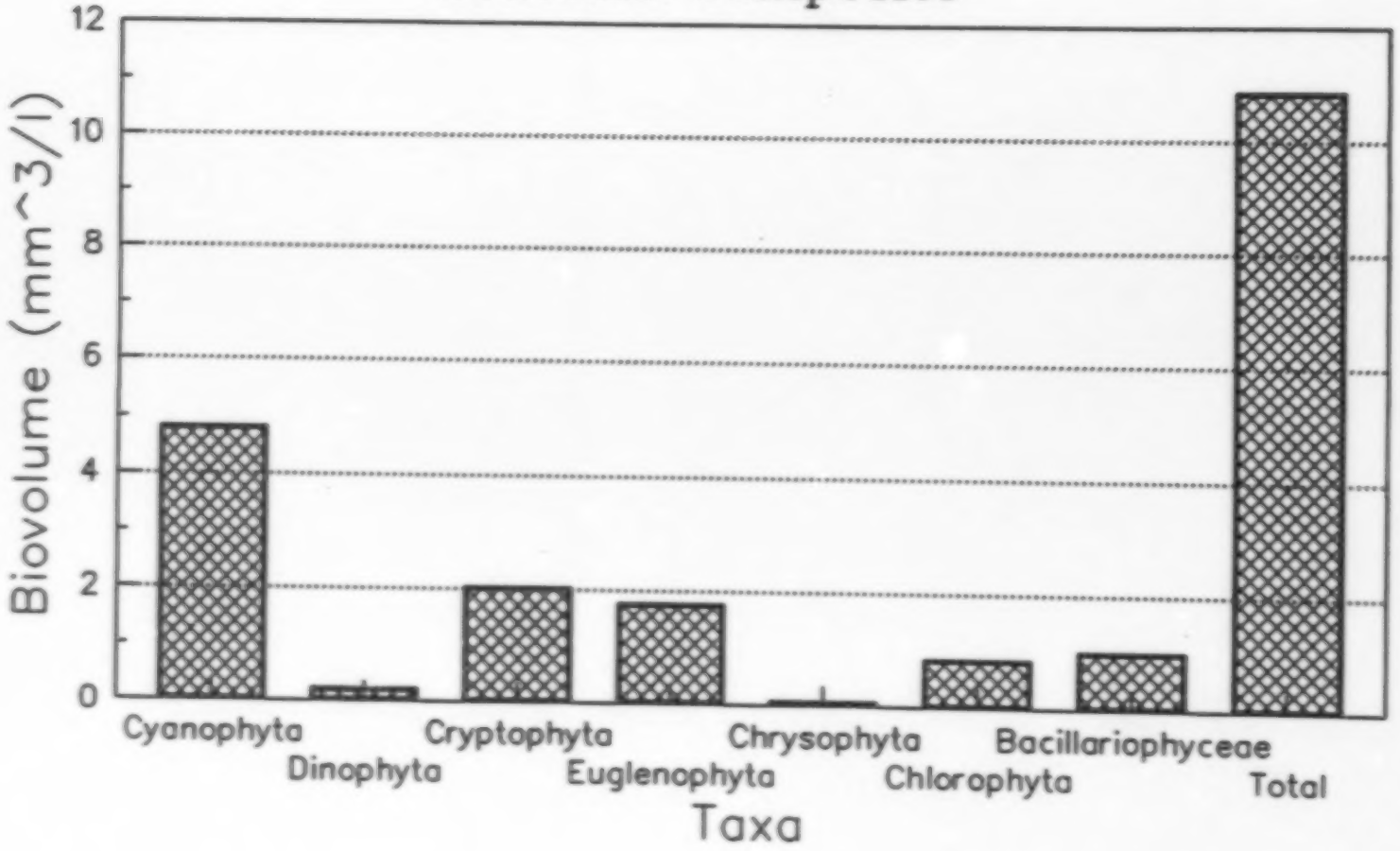


Fig. 35

Fig. 36

Fanshawe Reservoir
Phytoplankton Biovolume – 1989
Seasonal Composite



R OWN RISK
LE OBJECTS OF
IMMING AREA
EACH
NO SWIMMING IE
DUE TO ALGAE

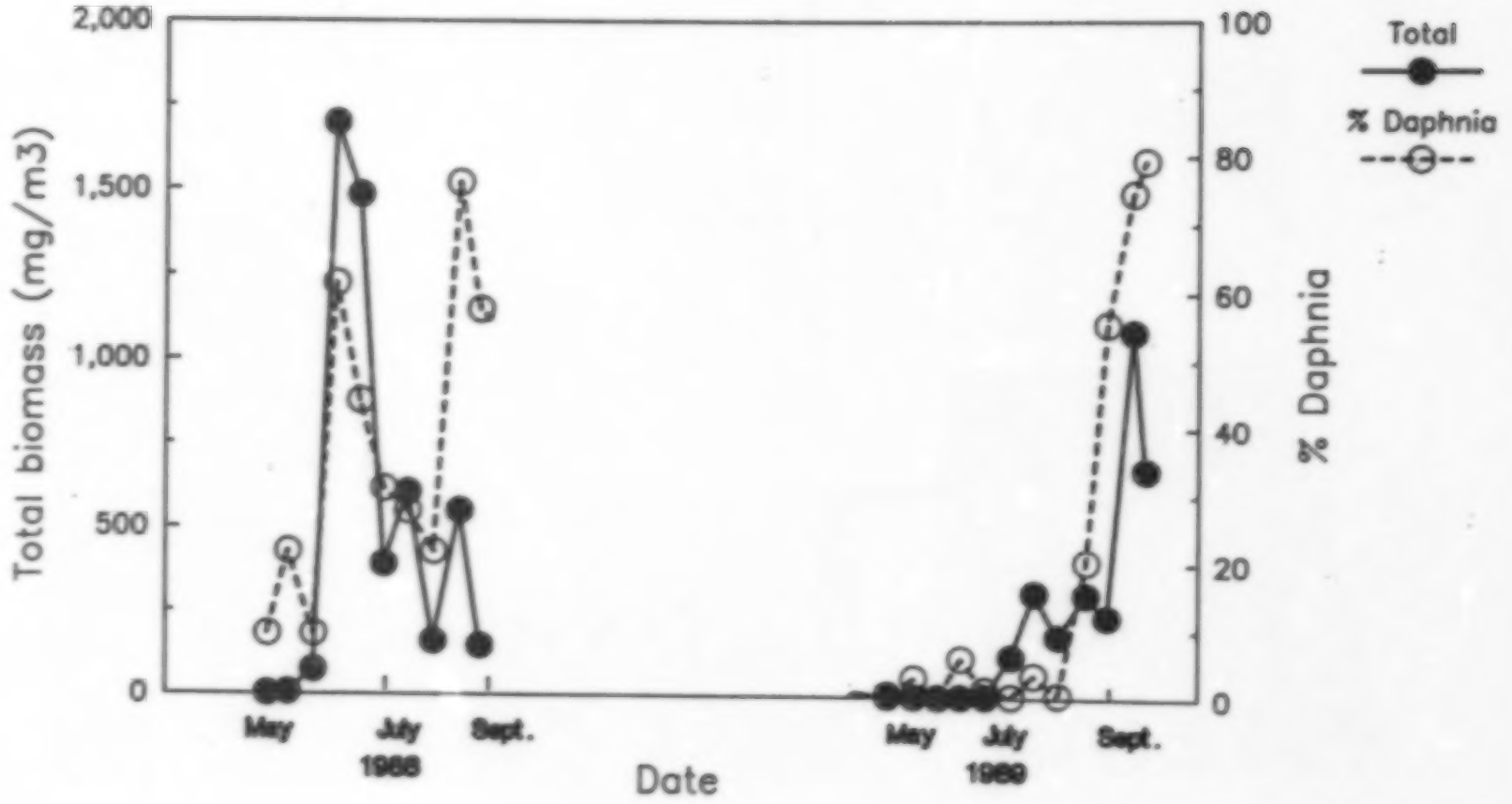
Fig. 37

Fig. 38

Fanshawe Reservoir

Zooplankton

1988 / 89



Fanshawe Reservoir

Zooplankton biomass as a % of total biomass

1988/89

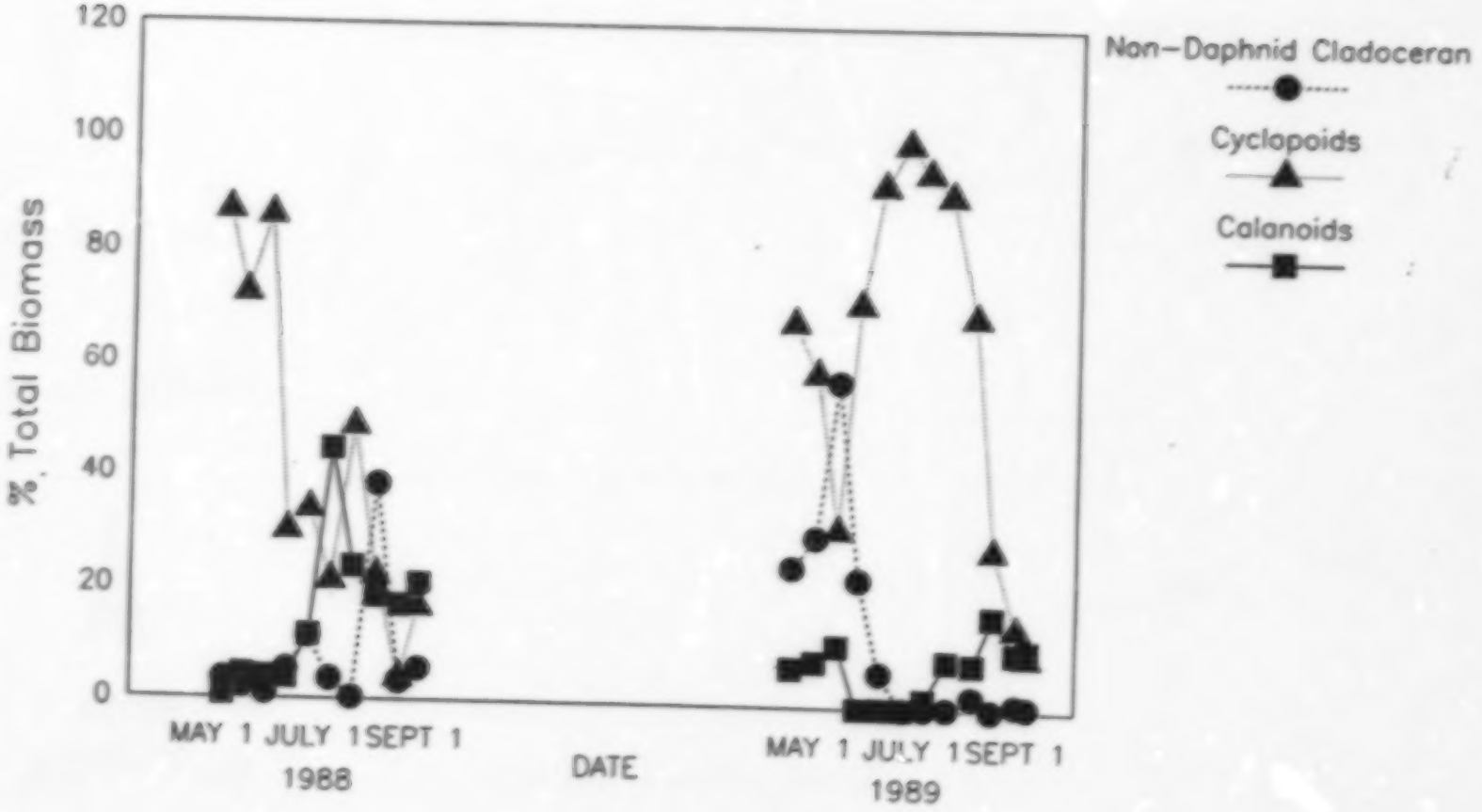


Fig. 19

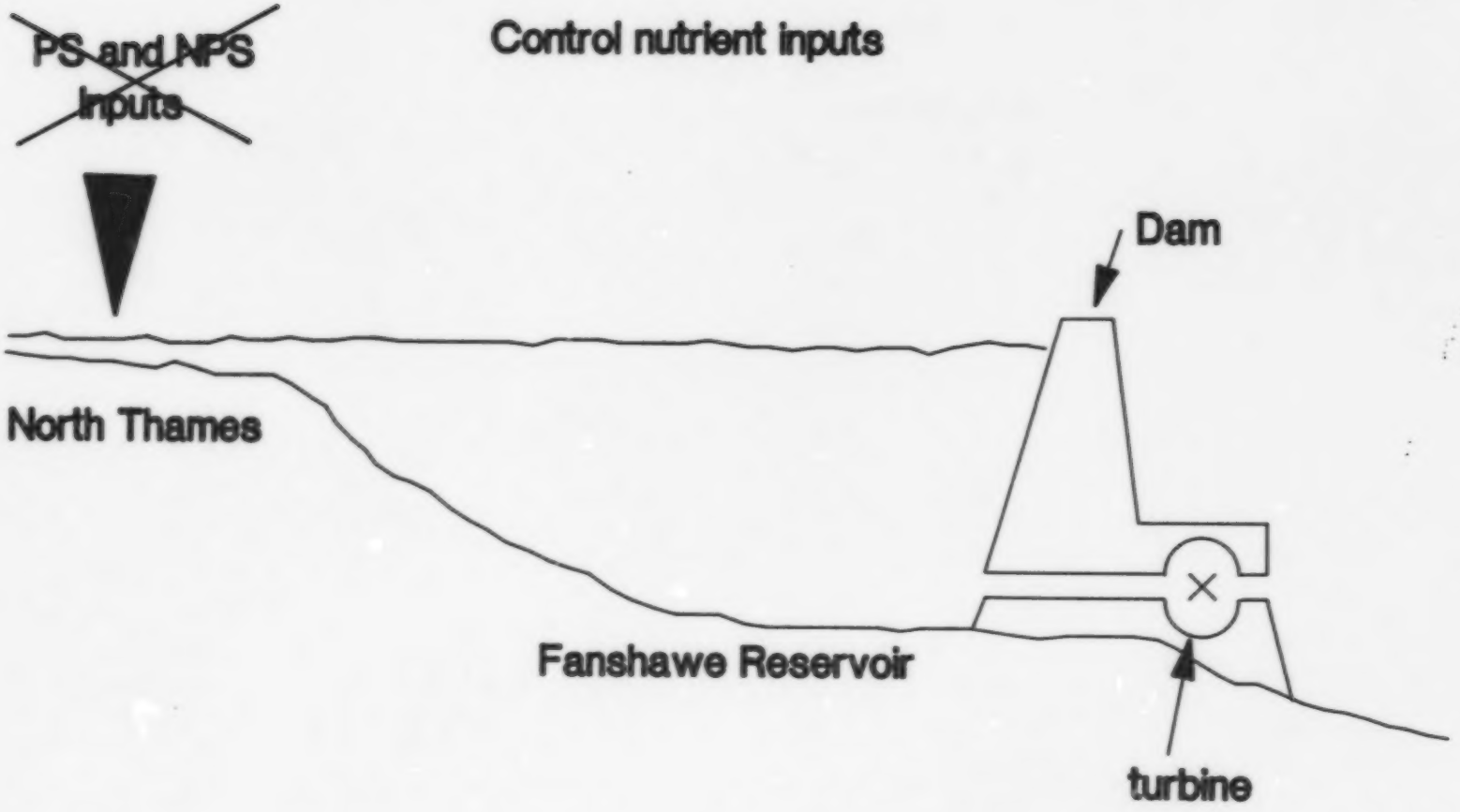


Fig. 40

sectional view of system

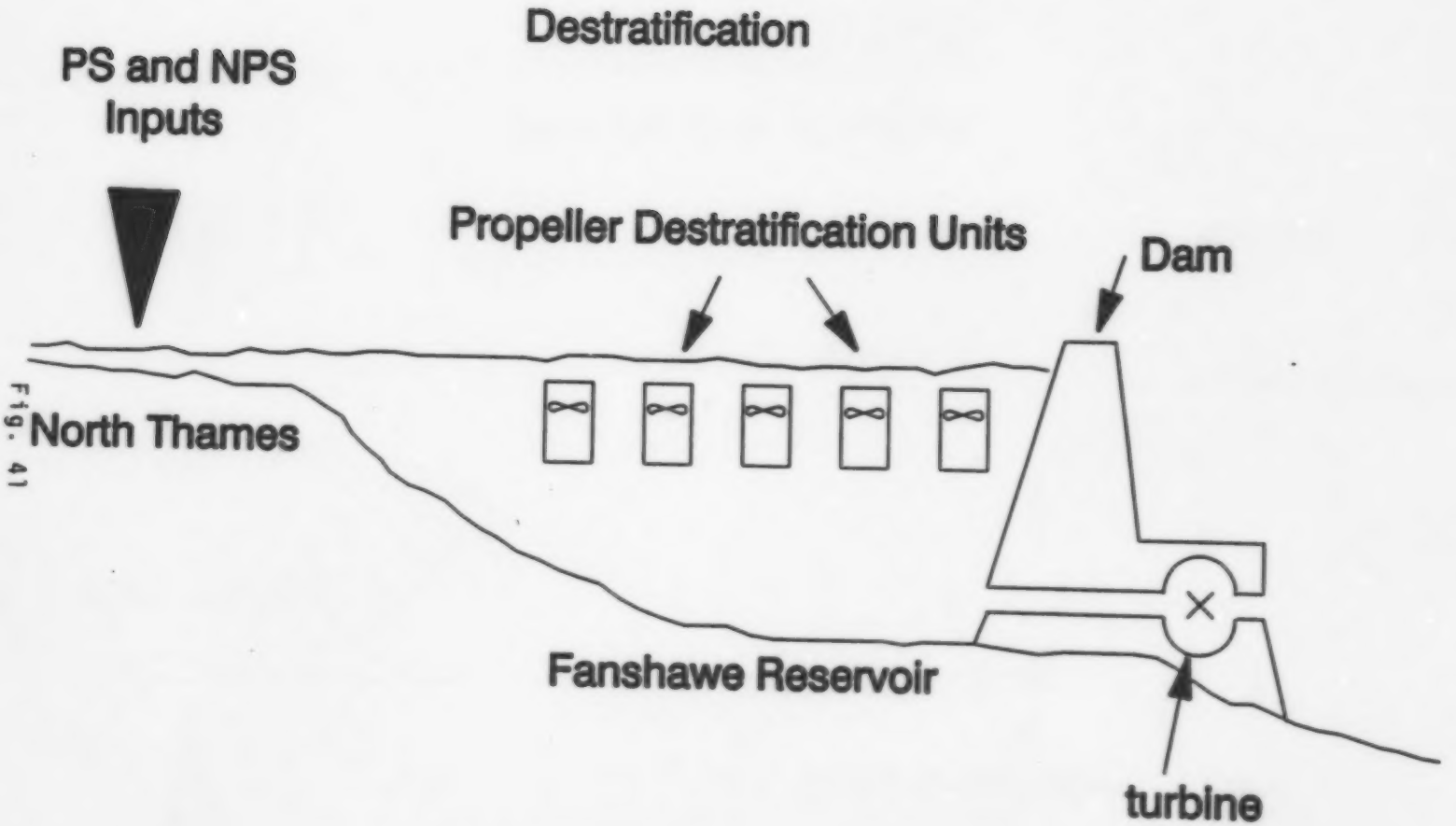


Fig. 41

sectional view of system

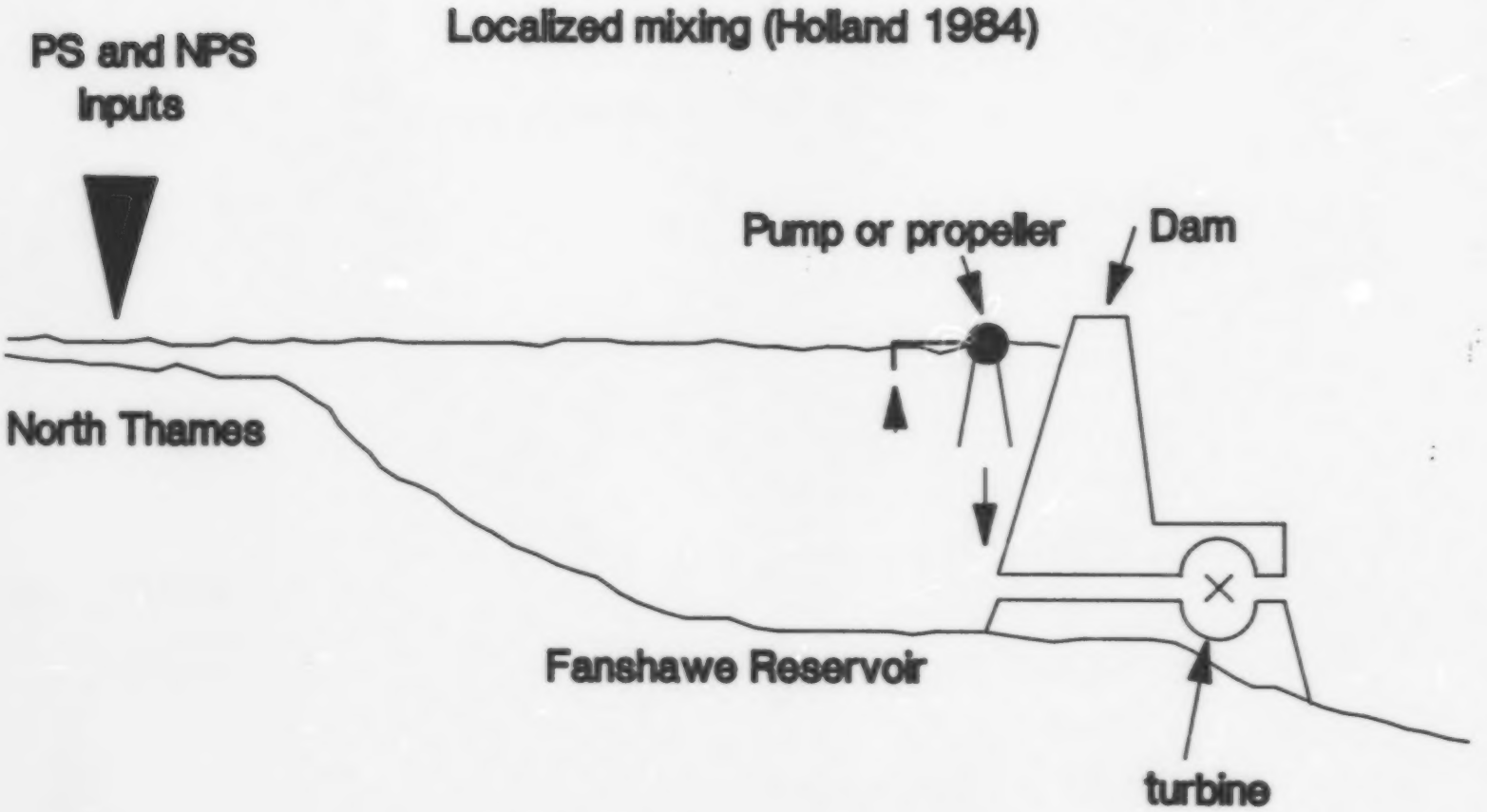
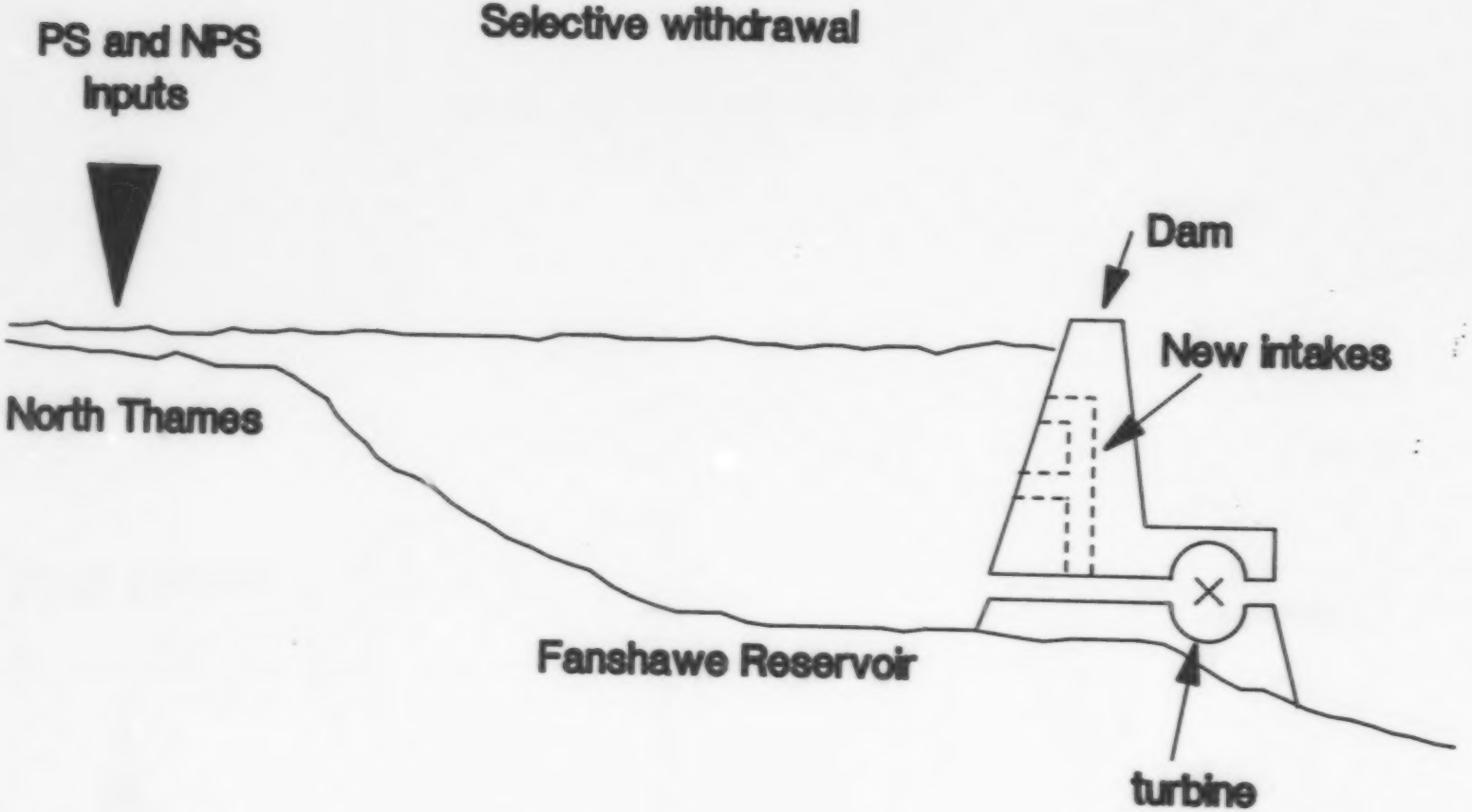
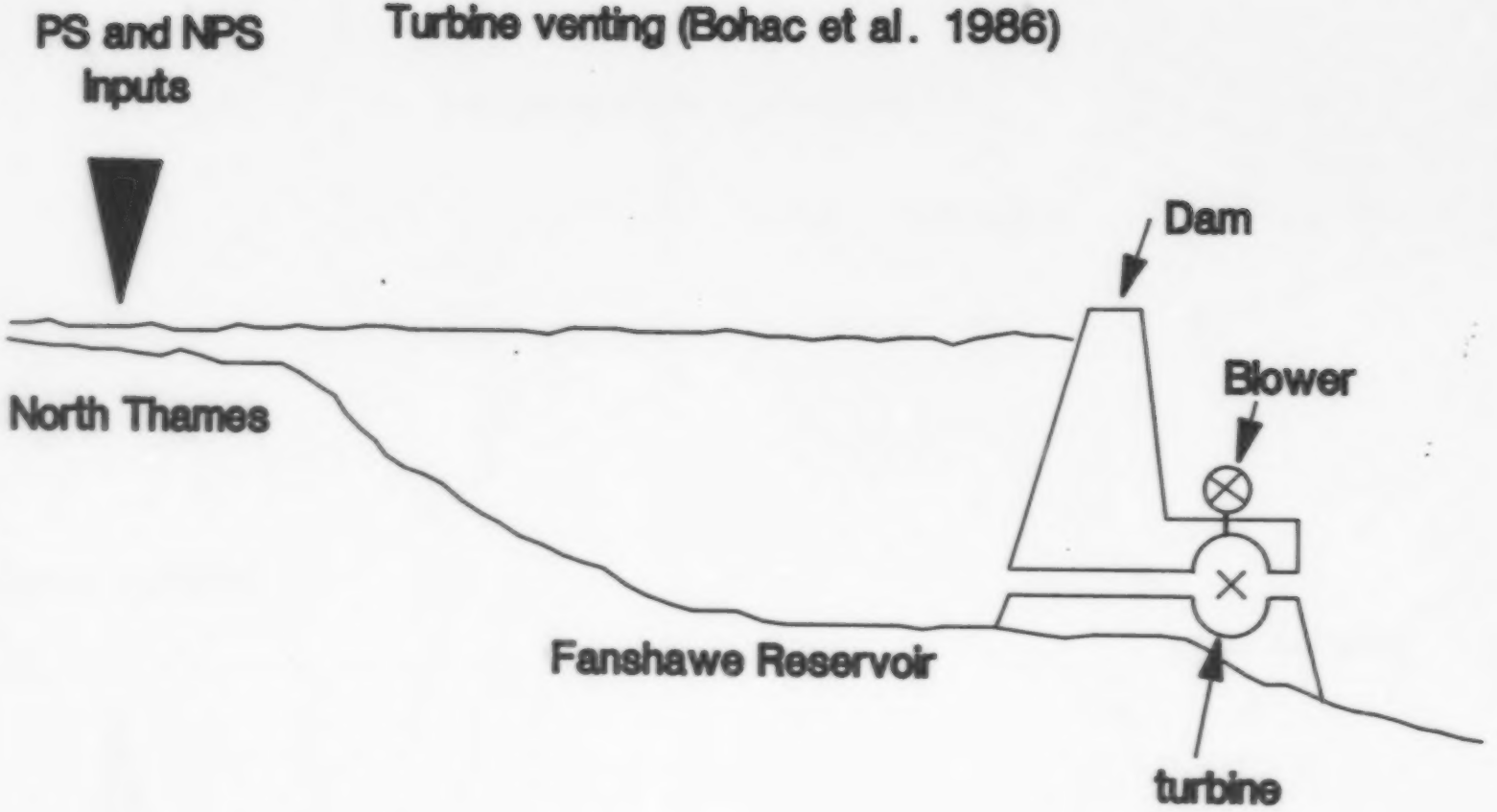

Localized mixing (Holland 1984)
PS and NPS
Inputs
Pump or propeller
Dam
North Thames
Fanshawe Reservoir
turbine
sectional view of system

Fig. 42



sectional view of system

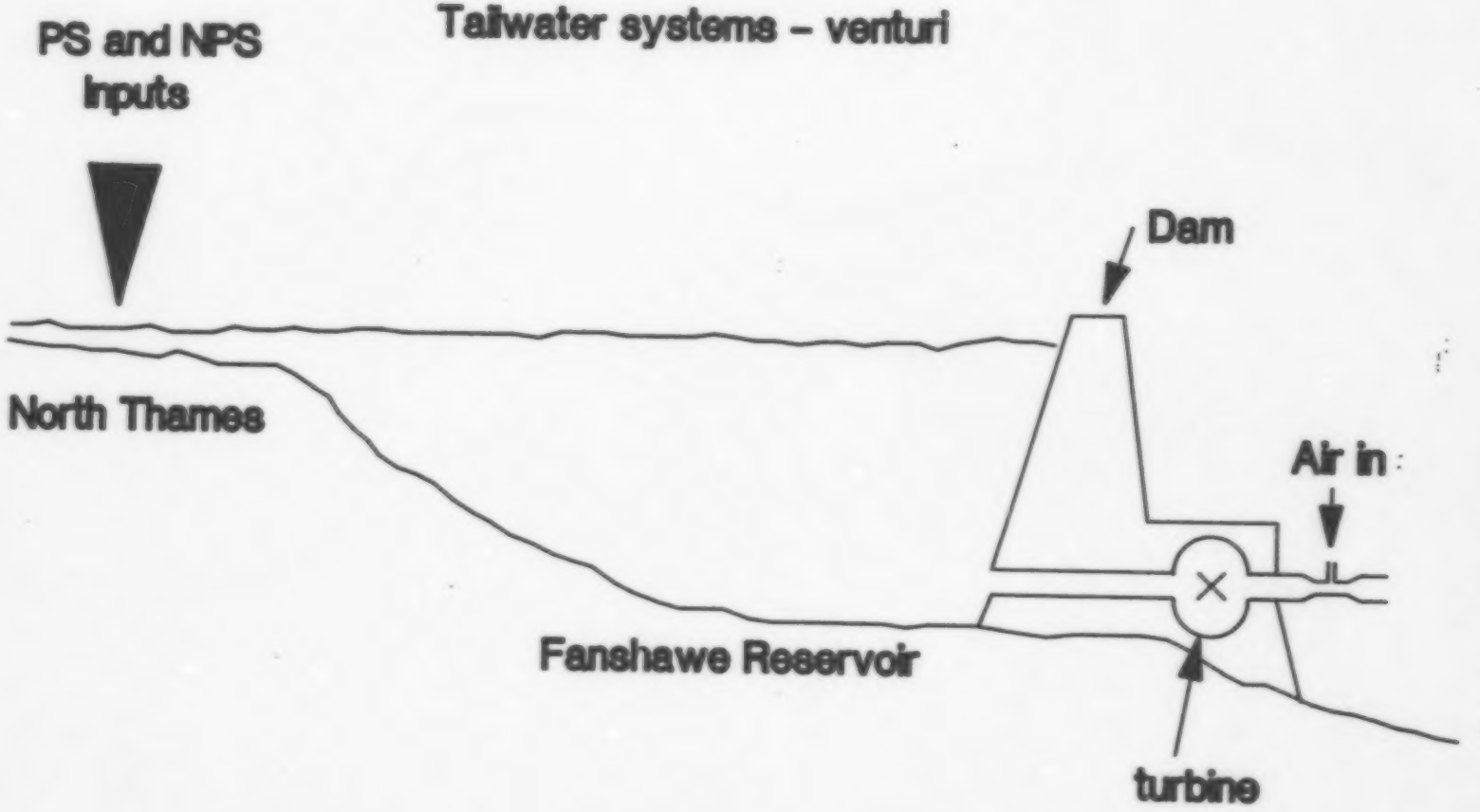
Fig. 43



sectional view of system

Fig. 44

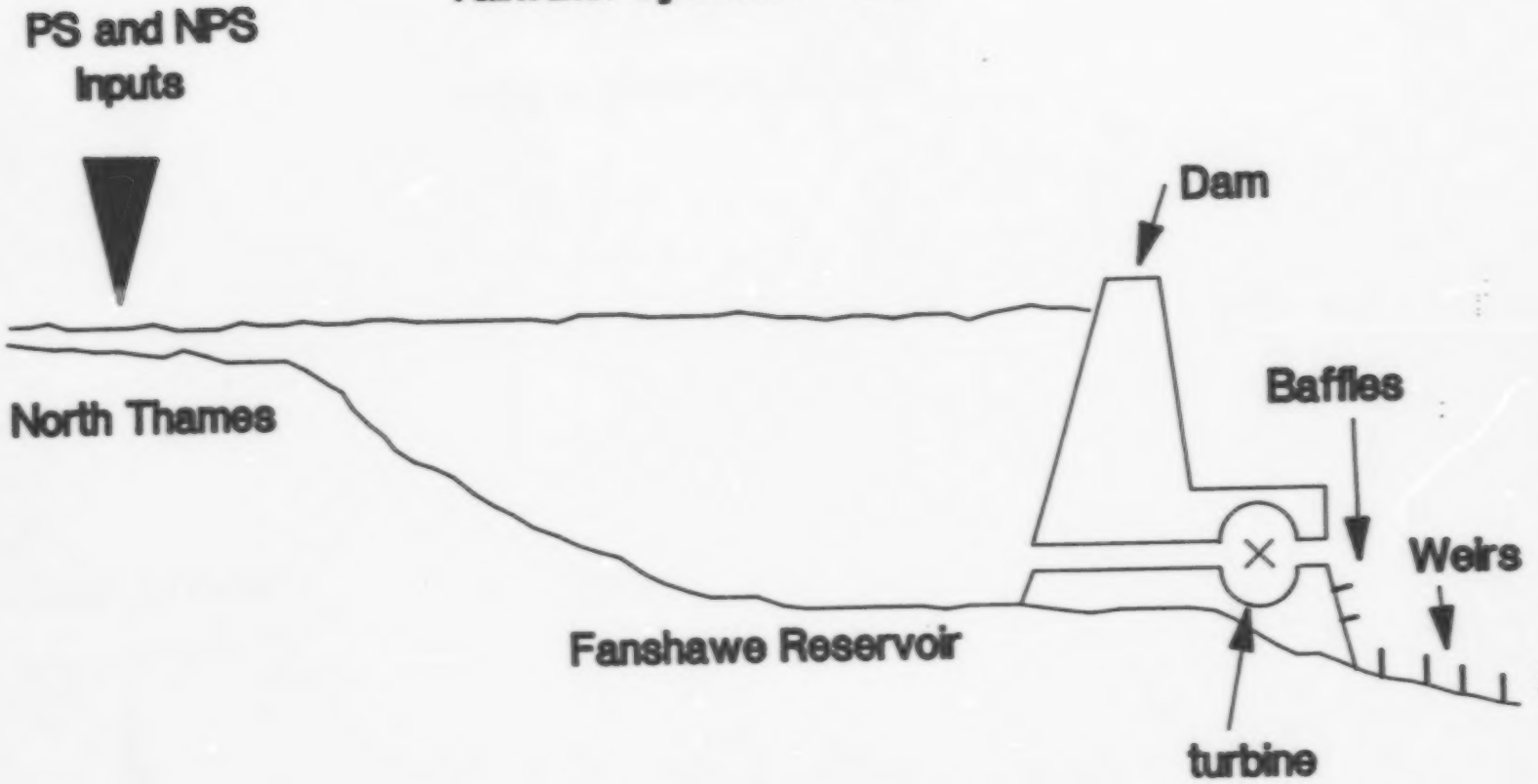
Tailwater systems – venturi



sectional view of system

Fig. 45

Tailwater systems – weir



sectional view of system

Fig. 46